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EXPOSURE ASSESSMENT COMPONENT OF THE
FIELD VERIFICATION PROGRAM:
OVERVIEW AND DATA PRESENTATION

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FOREWORD

In 1982, the U.S. Army Corps of Engineers (COE) and the U.S. Environmental Protection Agency (EPA) initiated a program to investigate three options for the disposal of material dredged from marine waterways. Two of these options, disposal in upland landfills and the creation of wetlands, were investigated by COE's Waterways Experiment Station (Vicksburg, MS). The third option, disposal in shallow coastal waters, was investigated by EPA's Environmental Research Laboratory - Narragansett (ERLN). One aspect of ERLN's program involved the description of the physical and chemical exposure field resulting from the disposal of dredged material at the Central Long Island Sound (CLIS) Disposal Site.

Initially, ERLN's exposure assessment efforts were directed towards the description of contaminant concentrations in time and space. This information was to be used in linking laboratory-based effects assessments with those carried out in the field. This report provides an overview of the exposure assessment component of the aquatic portion of the FVP, and describes the field sampling program and analytical methods used to obtain information concerning physical and chemical exposure conditions produced by the disposal operation and the resulting mound (source) of dredged material. The results of these analyses are presented and interpreted in this report within a descriptive framework. Included is information concerning physical and chemical characteristics of bottom sediments in and around the disposal mound, hydrographic parameters and chemical contaminant concentrations in the water column over the study area, and contaminant concentrations in tissues of selected infaunal and epibenthic organisms.

In addition to providing a description of exposure conditions resulting from shallow water disposal of dredged material, these data also provide insight into important physical and chemical processes controlling contaminant fate. The synthesis of this information in a process-oriented framework is the subject of a forthcoming series of manuscripts and reports. A major objective in the preparation of the present report was to provide the information necessary for this final synthesis.

ABSTRACT

The exposure assessment component of the aquatic portion of the Field Verification Program (FVP) related the source input of dredged material contaminants to corresponding concentration distributions in space and time in the vicinity of the disposal mound. The specific objectives for this component were: (1) to provide a description of the environmental exposure field at biological effects measurement stations in the water column (pelagic zone), in the sediments (benthic zone), and at the near-bottom/sediment-water interface (epibenthic zone); (2) to relate the source (dredge disposal mound) to near-field exposure measurements using process models for the vertical transport of contaminants and particulate materials; and (3) to determine the environmental processes controlling contaminant phase partitioning. This report provides an overview of the exposure assessment component of the aquatic portion of the FVP, and describes the methods used and results obtained from activities directed towards the first objective.

Approximately 55,000 cubic meters of material were disposed at the FVP site in the Central Long Island Sound (CLIS) Disposal Site between 26 April and 14 May 1983. This created a mound approximately 200 meters in radius in water 20 meters deep. The central part of the disposal mound was composed of sand-size particles mixed with silty material, while the fringes of the mound were composed primarily of fine-grained material. During the first few months after the disposal operation, the dominant sediment resuspension mechanism at the site was tidally-generated currents. There were no major storms during this period.

Water column chemical measurements indicated that there was a dredged material contaminant signal (3-7 times background) in the vicinity of the disposal mound for 1-2 months after the disposal operation. After this period of time, water column concentrations approached background levels. Tissue residues of caged mussels, deployed one meter above the bottom, gave similar results: elevated levels measured during the first few months after disposal were followed by a return to background levels thereafter. The contaminant signal in the sediments was more persistent, remaining elevated at mound stations throughout the 2.5-year postdisposal period of this study. It was also measurable (several times background) at a kilometer from the disposal mound. Moreover, polychaete tissue residues were elevated (generally two to three times background) at a kilometer from the disposal mound immediately following disposal, and remained above background for 1.5 years.

This report covers a period from 15 August 1982 to 30 January 1986 and the work was completed as of 30 January 1986.

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SECTION 1

EXECUTIVE SUMMARY

In 1982, the U.S. Army Corps of Engineers (COE) and the U.S. Environmental Protection Agency (EPA) initiated a multi-year study to evaluate the risks associated with three options for the disposal of dredged harbor material (Gentile et al., in press b). This program, designated the Field Verification Program (FVP), was designed to evaluate and field validate assessment methodologies for the prediction of environmental impacts associated with dredged material disposal in upland, wetland and aquatic environments. Evaluation of the impacts of aquatic disposal was conducted by EPA's Environmental Research Laboratory - Narragansett (ERLN).

The FVP provided a case study for the application of the hazard assessment research strategy developed at ERLN for ocean disposal (Bierman et al., 1986; Gentile et al., in press a,b). The underlying objective of this strategy is the development of quantitative relationships between source inputs of waste contaminants and their transport, transformation, fate and effects in the aquatic environment. A major component of hazard assessment consists of exposure assessment, which provides causal links between source inputs and subsequent environmental concentration distributions. Exposure assessment, then, involves the collection and synthesis of information concerning physical, chemical and biological processes which affect the transport, transformation and fate of waste contaminants. This information is essential for the prediction of environmental impacts resulting from ocean disposal of waste contaminants.

Within the context of hazard assessment, the exposure assessment component of the FVP had three primary objectives:

1. to describe the environmental exposure field at biological effects stations measured in the water column, in the sediments, and near the sediment-water interface;
2. to relate the contaminant source (disposal mound) to near-field exposure measurements using process models describing vertical transport of dissolved and particulate contaminants; and
3. to define and investigate environmental processes controlling contaminant phase partitioning as related to bioavailability and contaminant transport.

This report describes the methods used and data collected to meet the first of these objectives. The second and third objectives will be addressed in subsequent manuscripts and reports.

During late April and early May 1983, approximately 55,000 cubic meters of contaminated material dredged from Black Rock Harbor, a small industrialized harbor near Bridgeport, CT, were disposed at the FVP site in the Central Long Island Sound (CLIS) Disposal Area. These sediments contained high levels of trace metals (particularly Cd, Cu and Cr) and such organic contaminants as polynuclear aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and the pesticide Ethylan (Rogerson *et al.*, 1985). The disposal operation created a mound approximately 2 m in height and 200 m radius, with a thin veneer of dredged material extending outwards approximately 400 m (Scott *et al.*, 1984). The FVP site had been chosen to be as far removed from historic and on-going disposal operations as

possible within the confines of the CLIS Disposal Area. The mound, which resided at a depth of 20 m, was therefore considered to be the source of contaminants for all future work. Four stations were established along a transect oriented parallel to the direction of major tidal flow: CTR, located in the center of the disposal mound, and 200E, 400E, and 1000E, located 200, 400, and 1000 meters east of CTR. These latter stations were thus located on the flank, the fringe and off the mound, respectively. A fifth station, REFS, was established approximately 3 km south of CTR and well outside of the disposal area. This station was regarded as the control site for subsequent exposure assessment field work.

Information necessary to meet the first objective outlined above was gathered by grab sampling in the water column and sediments, ship-board collection of vertical hydrographic profiles, and deployment of in situ instrumentation. Sediment exposures were determined by physical and chemical analyses of samples collected on a monthly or seasonal basis, and through benthic camera surveys. Water column exposures were determined by sampling at various horizontal and vertical stations in the water column, on time scales ranging from hours to seasons. Additional information was obtained through the deployment of in situ instrumentation which measured hydrographic parameters and suspended solids concentrations on a near-continuous basis. Exposures near the sediment-water interface were characterized through deployment of a second in situ instrument which measured suspended solids concentrations at approximate intervals of 1 cm from the sediment surface to 1 meter above bottom.

Physical parameters measured in the water column included temperature, salinity, dissolved oxygen (DO), light attenuation, and suspended solids concentration. Eight trace metals, 52 individual organic contaminants and classes of contaminants, including PAHs, PAH related compounds, and PCBs, and total

carbon were measured in sediment samples as well as in the water column. Water column concentrations were further partitioned into dissolved and particulate fractions. Contaminants were also measured in an indigenous infaunal polychaete worm (Nephtys incisa), and in blue mussels (Mytilus edulis) deployed in cages at 1 meter off the bottom. Both of these species were used in biological effects experimentation in the field and the laboratory (Phelps et al., in press).

Water column temperature and salinity displayed seasonal cycles, with temperatures ranging from 0 to 25°C and salinities ranging from 23 to 29 ppt. Salinity was strongly influenced by patterns of freshwater runoff from Connecticut river systems, which reached peak flow during the spring melt in March through June. Vertical stratification of both parameters developed in March-April, such that surface waters were 3-4°C warmer and 2 ppt less saline than bottom waters. These differences persisted through late summer, when fall turn-over resulted in more or less vertically uniform temperature and salinity profiles. Summer water column DO concentrations also displayed vertical stratification as a result of increased oxygen consumption in bottom waters by benthic respiration and decomposition. The pycnocline present during the summer months acted as a barrier to downward movement of DO from oxygen-rich surface waters. The mound of dredged material had no apparent effect on DO concentrations. Variation in all hydrographic parameters was also associated with the movement of heterogeneous water masses in the Sound. This variation was particularly evident in the near-continuous records of temperature and salinity obtained from in situ instrumentation. In general, variation in temperature, salinity and DO between stations was no greater than that occurring at single stations over the course of a day. No major storms occurred during the summer and early fall following disposal.

Background suspended solids concentrations at 1 meter above bottom were typically on the order of 5-25 mg/l. Turbidity in summer decreased to a minimum at the depth of the pycnocline, and increased near the surface due to primary production. Tidally induced current speeds measured by in situ instrumentation deployed at 1 meter above the bottom were strongest along the east-west axis, approaching 0.6 knots, while those measured in the north-south direction approached 0.4 knots. These currents apparently resulted in tidal scouring and resuspension of bottom sediments in a cyclic fashion. Suspended solids concentrations also displayed sharp, aperiodic excursions from background up to 45-50 mg/l. Possible explanations for this phenomenon include the passage of turbidity plumes produced by disposal operations, intense resuspension events produced by the activities of benthic organisms and commercial fishery operations, and temporary blockage of the transmissometer optical windows by floating debris. The frequency of these excursions increased during the disposal operation.

Elevated near-bottom water column contaminant levels 3 to 7 times background were observed over the mound for up to 1-2 months following disposal, after which time concentrations fell to background levels. Large signals were measured near bottom only, and are thought to be associated with tidal resuspension of the unconsolidated dredged material shortly following disposal as well as to residual suspended material from the disposal operation itself. Increased contaminant concentrations were detected as far away as 3 km at the REFS control station, although these most likely resulted from the disposal of over 1 million cubic meters of material at other sites in the CLIS Disposal Area. Partition coefficients calculated for organic compounds and trace metals agreed well with reported literature values. In general, the water column exposure field was particularly dynamic, and rather poorly characterized by the monthly sampling scheme.

Prior to the disposal operation, sediments collected around the designated site were physically and chemically uniform. Postdisposal CTR and 200E sediments were more coarse grained than those collected at other stations or than predisposal sediments. This is thought to be due to particle sorting as a result of different settling times, to subsequent tidal scouring of the mound, and to enrichment by clean sand used in stabilizing barge loads during transport. Gravity cores collected at CTR indicated large vertical variability in grain size, with fine material sandwiched between layers of sand and relic clay clasts. These patterns of particle size were reflected in contaminant levels across stations. CTR sediments were found to be extremely heterogeneous with depth, and in general less contaminated than BRH material collected for laboratory testing prior to dredging. The 200E site displayed more uniform contaminant levels with depth, with values similar to those at CTR. The veneer of BRH material at 400E was clearly visible as a thin layer of contaminated sediment overlying clean CLIS background sediment. Increased contaminant levels were also observed at 1000E. Although water-borne contaminants may have been deposited at this site during and after the disposal operation, evidence suggests an errant release of dredged material approximately 50 m northwest of this station (Feng and DeGoursey, 1984). Sediments at REFS remained generally unaffected. Total carbon concentration was also elevated in mound sediments, and was correlated with organic contaminant levels at 200E, but not at CTR. The lack of correlation at CTR is somewhat puzzling, but may be attributable to the heterogeneous and coarse-grained nature of those sediments. Surficial sediments (approximately 0-4 cm depth) became less contaminated with time, although strong signals were still observed 2.5 years after disposal. Deeper sediments did not display this loss of contaminants.

Tissue residues in Mytilus and Nephtys reflected

contaminant levels in the water column and sediments, respectively. Mussels deployed at all stations (including REFS) showed increased concentrations (up to 3 times background) for 2-3 months following disposal, but returned to predisposal levels thereafter. Uptake was strongest at CTR, and decreased with distance along the transect. The elevated REFS levels are thought to be due to disposal of dredged material at sites other than the FVP, as elevated levels of Ethylan and some PAHs were observed prior to the FVP operation. Nephtys collected from sediments at 400E and 1000E showed increased contaminant levels (up to 20 times background) which persisted 1.5 years after disposal, though 1000E worms displayed near-background levels by that time. Following burial during the disposal operation at CTR, worms were again collected at this station in spring of 1984. These showed the highest levels of tissue residues of all worms collected at that time, and remained 3-4 times background through fall of that year. No evidence of increased levels in REFS worms was observed during the study.

In addition to providing information necessary to replicate field exposure conditions in the laboratory, data presented in this report will be used to develop and test process-oriented models for description of contaminant fate and transport in marine systems. The results of these syntheses will better enable environmental scientists to predict the effects of ocean disposal on marine communities and human health.

SECTION 2

INTRODUCTION

The U.S. Army Corps of Engineers (COE) and the U.S. Environmental Protection Agency (EPA) jointly conducted a comprehensive study, referred to as the Field Verification Program (FVP), to evaluate the risks associated with various disposal options for dredged materials (Gentile et al., in press b). The approach used in the FVP was to evaluate and field validate assessment methodologies for predicting the environmental impacts of dredged material disposal in aquatic, upland, and wetland environments. The EPA Environmental Research Laboratory - Narragansett (ERLN) conducted the aquatic portion of this program.

The ERLN research plan for the aquatic portion of the FVP was an application of the risk (formerly referred to as hazard) assessment research strategy that has been developed at ERLN for ocean disposal (Bierman et al., 1986; Gentile et al., in press a,b). The basic objective of this strategy is the development of quantitative relationships between source inputs of waste contaminants and their transport, transformation, fate, and effects in the aquatic environment. Risk assessment consists of two basic components: exposure assessment and effects assessment. Exposure assessment provides the causal link between source inputs and environmental concentration distributions, while effects assessment provides the causal link between exposure concentrations and biological effects. Synthesis of results from these two components provides a unified framework for assessing the response of an environmental system to the

disposal of a particular waste material.

Exposure assessment involves the collection and synthesis of information on source inputs with information on the physical, chemical, and biological processes which affect transport, transformation, and fate of waste contaminants in the environmental system. Mathematical models provide a useful quantitative framework for this synthesis. Such models can describe physical processes affecting contaminant movement in the system, such as convection and dispersion in the water column, and mass fluxes between the water column and the underlying sediments. They can also describe chemical and biological processes affecting transformation and distribution of contaminants in the water column, sediment, and biota. These processes include adsorption/desorption, degradation, and biological uptake.

This report describes the exposure assessment component of the aquatic portion of the FVP, its specific objectives, and the data acquisition and synthesis necessary to meet the overall objectives of the FVP (Gentile et al., in press b). Data acquired as part of the study, and presented in this report, cover predisposal conditions, as well as conditions up to 2 years after disposal. A series of manuscripts and reports to be published later will describe the synthesis conducted with these data to investigate the causal links between the source input of dredged material and the resultant environmental exposures.

RATIONALE

Since EPA is a regulatory agency, research conducted at ERLN must relate to specific needs in support of the ocean disposal regulatory process (Figure 1). The endpoint is the potential ecological impact which may result from input of contaminants into the environment. Environmental exposure is the

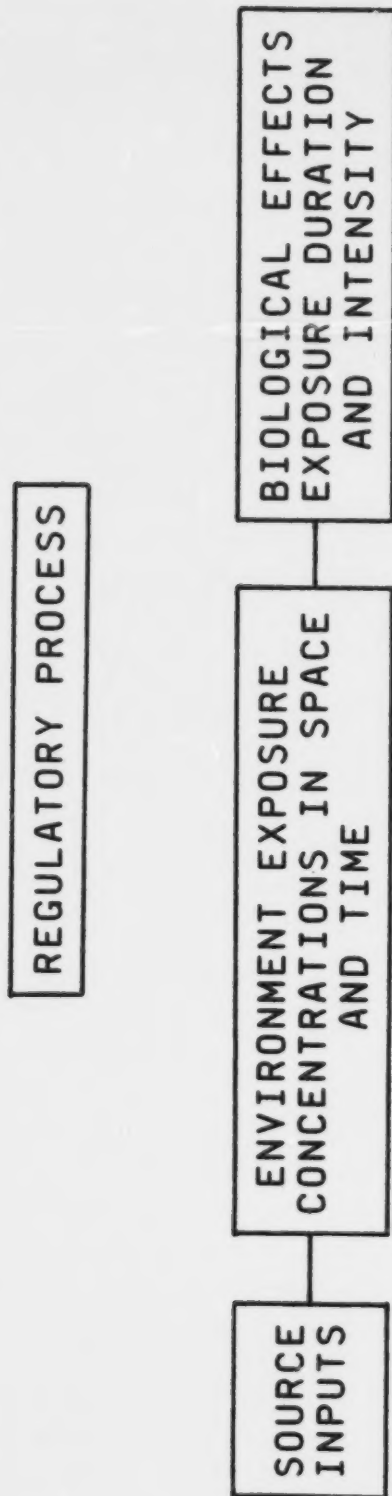


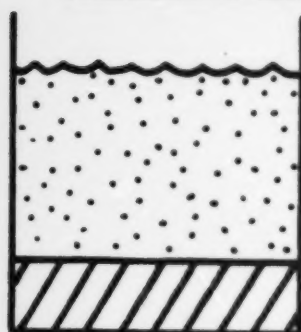
Figure 1. Regulatory process for ocean disposal.

linkage between input of the material into the environment and subsequent biological effects.

One of the main thrusts of the FVP was the field validation of laboratory assessment methods for predicting environmental impacts of dredged material disposal (Gentile et al., in press b). Laboratory tests have been conducted with both suspended and solid phase material from Black Rock Harbor (BRH), the source of the dredged material, in flow-through, constant exposure laboratory systems (Phelps et al., in press). Further laboratory tests are currently being conducted to simulate actual field exposure conditions observed at the center of the disposal mound, on the fringes of the mound, and at locations away from the mound (Figure 2). The same biological effects measurements were conducted at the FVP site (Phelps et al., in press).

There are different levels at which the laboratory-to-field linkage could be made for biological effects measurements. One such level would be the comparison, or correlation, of laboratory effects results conducted under standard exposure conditions with effects measurements obtained from the field. A positive correlation could be fortuitous because the degree of comparison between laboratory and field exposure conditions would be unknown. Further, the results obtained from this level of comparison would not be applicable to any other disposal site. Another level at which this link could be made would be the comparison of laboratory effects results conducted under simulated field exposure conditions with effects measurements obtained from the field. Positive or negative correlations could then be related to the ability of the laboratory tests to predict what occurs in the field. This latter approach was taken in the aquatic portion of the FVP. The exposure assessment component of the FVP provided the information necessary to simulate field exposure conditions in the laboratory, and also provided estimates of spatial and temporal variability of

EPA - COE
FIELD VERIFICATION PROGRAM



**LABORATORY
EXPERIMENTAL CONDITIONS**

**BRH DREDGED MATERIAL
CONTROL SEDIMENT**

- SOLID
- SUSPENDED



CENTRAL L.I. SOUND DISPOSAL SITE

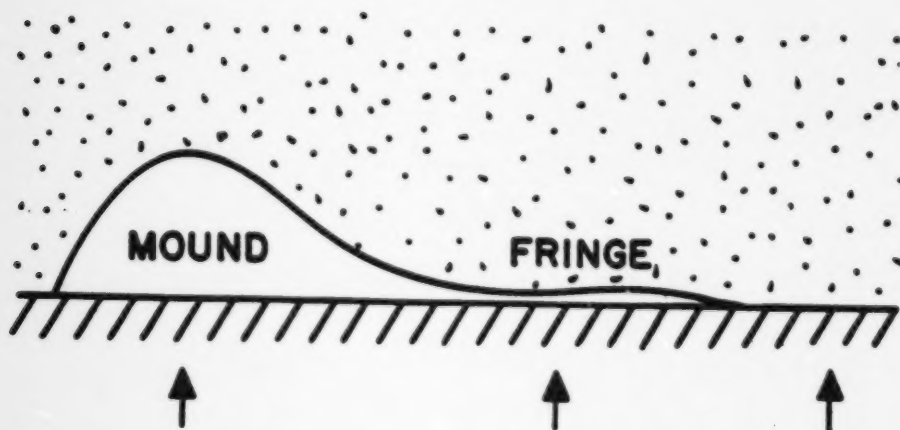


Figure 2. Field Verification Program laboratory to field link.

controlling environmental parameters in the vicinity of the disposal site. In addition, this information has been used to develop and validate process models for vertical transport and contaminant phase partitioning at the disposal site (the subject of subsequent manuscripts and reports). These models provide a basis for extending the exposure assessment results from this study to other disposal sites.

OBJECTIVES OF THE EXPOSURE ASSESSMENT COMPONENT

The general objective of the exposure assessment component of the aquatic portion of the FVP was to develop and field validate quantitative assessment methods for relating dredged material inputs to the transport, transformation, and fate of associated contaminants. The specific objectives were:

1. to determine the environmental exposure field at the biological effects measurement stations in the water column (pelagic zone), in the sediments (benthic zone), and at the near-bottom/sediment-water interface (epibenthic zone) using a combination of field sampling and in situ instrumentation;
2. to relate the contaminated source (disposal mound) to near-field exposure measurements using process models for the vertical transport of contaminants and particulate materials; and
3. to determine the environmental processes controlling contaminant phase partitioning, as related to bioavailability and contaminant transport and fate.

A discussion of how the above objectives were met is best centered about the conceptual model we used to describe the movement of contaminants and particulate material in shallow

aquatic systems (Figure 3). The vertical compartments in Figure 3 correspond to the various zones in which the effects assessment measurements were conducted. The dredged material on the disposal mound, along with the associated contaminants in the particulate and dissolved phases, was considered to be the source input. This material could be resuspended by a combination of various processes, such as tidal currents and wind-generated waves. Repartitioning of the contaminants between the dissolved and particulate phases could result as the dredged material mixed with background material. This material could move vertically in the water column and be transported horizontally by water motion. Subsequently, the material could be deposited and again resuspended and transported horizontally. Contaminants could continuously repartition between the dissolved and particulate phases in this process. The consequence of these processes would be an exposure field which varied continuously in space and time in the vicinity of the disposal site. It was necessary, at a minimum, to quantify this exposure field in a manner compatible with the information requirements of the effects assessment field work and laboratory simulations.

The field sampling plan, discussed in the next section, was designed to measure the environmental exposures where biological effects assessment measurements were being conducted. Sediment exposures were determined by analyses of sediment cores collected at various horizontal locations and at time periods of months to seasons. Water column exposures were determined by (1) field sampling for contaminant and suspended solids concentrations at various horizontal stations, at different vertical levels in the water column, and at time periods of months to seasons, and (2) continuous deployment of in situ instrumentation developed by the University of Connecticut (Bohlen, 1982) for determination of suspended solids concentrations one meter above the bottom at a single station,

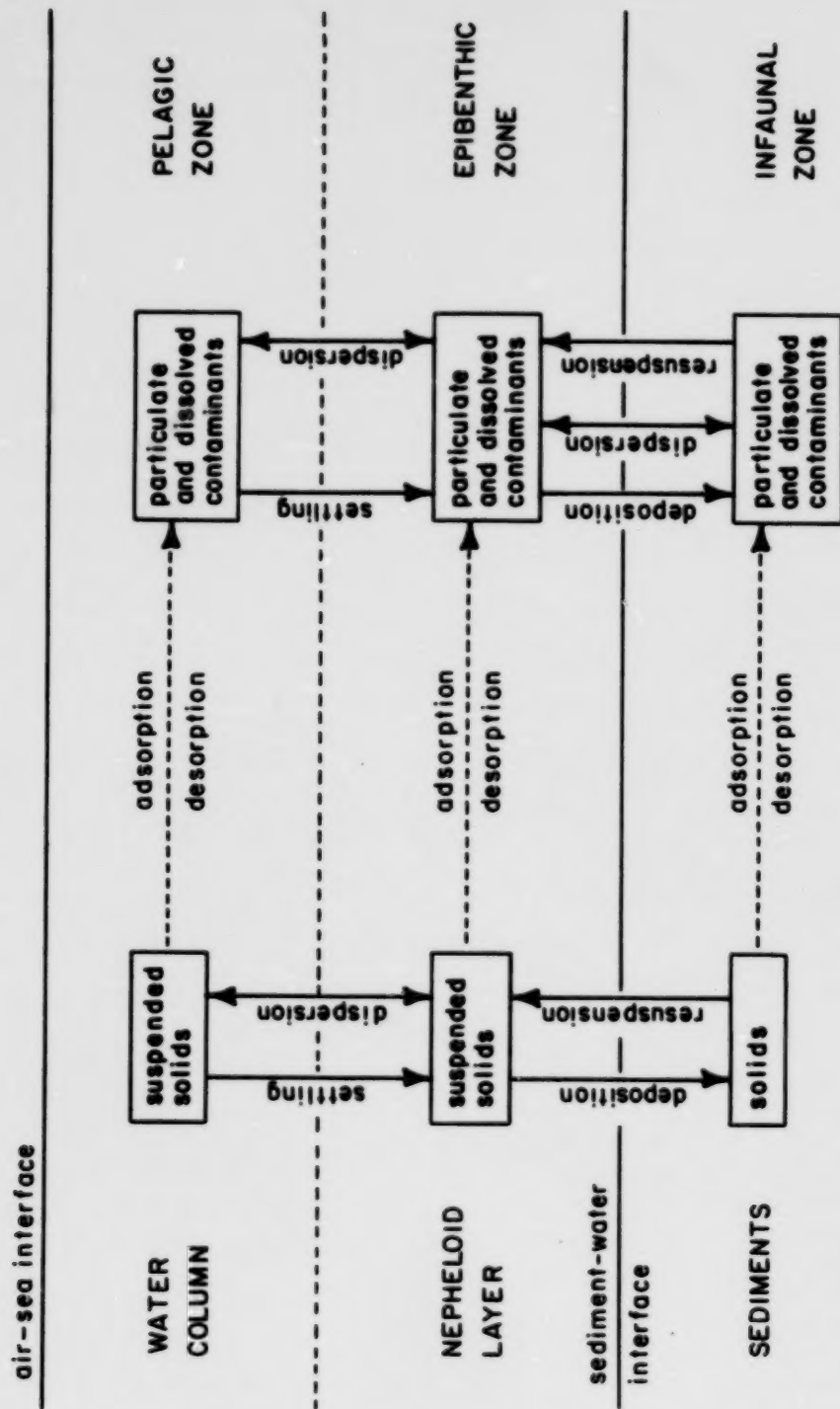


Figure 3. Schematic diagram of conceptual model for the exposure assessment component of the FVP.

with burst sampling at 15 minute intervals. One meter above the bottom corresponds to the deployment depth of caged mussels used as one biological measure in the effects assessment field work (Phelps et al., in press). Near-bottom/sediment-water interface exposures were determined by a second in situ instrument, an acoustic profilometer developed by Ohio State University (Bedford et al., 1982). The acoustic profilometer measured suspended solids concentration profiles from a location one meter above the bottom to the sediment-water interface, at approximately one cm intervals. This provided field exposure information for organisms that live near the sediment-water interface.

Models for vertical transport of material were developed from laboratory flume experiments at the University of California, Santa Barbara, to account for both physical and biological processes which affect the flux of material across the sediment-water interface (Lick, 1982, 1984; Tsai and Lick, 1985a). These laboratory experiments were validated with data obtained by the in situ instrumentation described above and by benthic surveys conducted with the REMOTS® benthic camera (Rhoads and Germano, 1982). The linkage between laboratory and field resuspension results was further strengthened through the use of a device that has been developed for measuring sediment resuspension parameters in the field (Lick et al., 1984; Davis et al., 1984; Tsai and Lick, 1985b). The results of this work will be reported elsewhere (Davis et al., in prep.).

In marine systems, the transport and bioavailability of contaminants are determined in part by their degree of association with particulate matter. Laboratory experiments were conducted at Manhattan College (Di Toro and Horzempa, 1982; Di Toro et al., 1982, in press; Di Toro and Mahony, 1985) to develop process models for phase partitioning and adsorption/desorption of BRH contaminants under conditions at

the FVP site in both the water column and sediments. Emphasis was placed on the degree of reversibility of contaminant desorption, and on the effects of suspended solids concentration on the observed partition coefficient. Field sampling analyses for BRH contaminants were used to validate these process models.

DESCRIPTION OF DATA ACQUISITION

A summary of the exposure assessment field data is given in Table 1. There were two appropriate sets of scales for this project. The first corresponded to the scales of interest for effects assessment measurements, that is, the time frame over which biological effects tests were conducted and the regions which pelagic, epibenthic, and infaunal organisms inhabit. Measurements made on these scales were actually bulk measurements because they do not contain information on processes which occur at smaller scales, but rather reflect an integrated measure of what occurs on these smaller scales.

The second set involved the scales at which controlling dynamic processes actually occur. It has become evident only in the past few years that these processes, such as those controlling interactions at the sediment-water interface, occur on scales which are quite small. For example, Heathershaw (1976) determined from measurements in the Irish Sea that as much as 70% of turbulent energy production in the bottom boundary layer occurs in about 5% of the total time of record, with event durations of 5-20 seconds. This means that physical processes which resuspend sediment at the bottom do not act continuously, but occur as intense bursts which are randomly spaced in time. Only when measurements are made on scales over which the actual processes occur can one understand what bulk measurements really mean, and then develop a correct parameterization of these processes for use at the bulk scale level.

TABLE 1. FIELD AND LABORATORY DATA ACQUIRED FOR THE EXPOSURE ASSESSMENT COMPONENT OF THE FIELD VERIFICATION PROGRAM

	Water Column	Sediment	Biota
Physical	in situ instruments: turbidity, temperature, salinity, currents, waves benthic camera imagery laboratory flume experiments: resuspension parameterization cruise data: suspended solids, temperature, salinity, DO tides water levels meteorological information satellite imagery disposal records	side scan sonar: bathymetry bulk analysis: grain size, water content diver observations benthic camera imagery laboratory flume experiments: sediment resuspension parameterization box cores bottom shear stresses: from in situ instrumentation	effects of biota on physical properties of sediments
Chemical	metals organics partition coefficients	metals organics total carbon	pelagic epibenthic infaunal
Biological		benthic camera imagery box cores diver observations	

Another consideration of scale for the acquisition of data involved variability in the actual environmental system. The field program was designed to measure variability that occurs at different time scales: hourly, tidal, monthly, and seasonal. Vertical variability in the water column was investigated by measuring selected parameters at one-meter intervals. Horizontal variability was investigated by placing sampling stations along perpendicular transect lines centered on the disposal mound (Phelps et al., in press) as shown in Figure 4. The biological effects measurement stations were located at the center of the mound (CTR), 200 meters east of CTR (200E), 400 meters east of CTR (400E), 1000 meters east of CTR (1000E), and at a reference station (REFS) located south of the disposal area. The 200E station is located on the flank of the mound, 400E is on the fringe of the mound, and 1000E is a kilometer away in the predominant tidal direction. REFS is located approximately 3 km to the southwest of CTR, and was used as a control station in this study. Additional stations were sampled at locations around the mound so that parameters could be measured as the tidal flow came over the mound and transported material into the surrounding area.

DESCRIPTION OF DATA SYNTHESIS

The emphasis during the initial phase of the exposure assessment component of the FVP (the subject of this report) was on the description of field exposure conditions for pelagic, epibenthic, and infaunal organisms. This information was required to conduct laboratory effects assessment tests under simulated field exposure conditions. The basic parameters required to do this were suspended solids concentrations and the particulate, dissolved, and bedded sediment concentrations of contaminants present at the disposal site. In addition, information was required on ambient environmental conditions and their natural variability. This report provides a summary

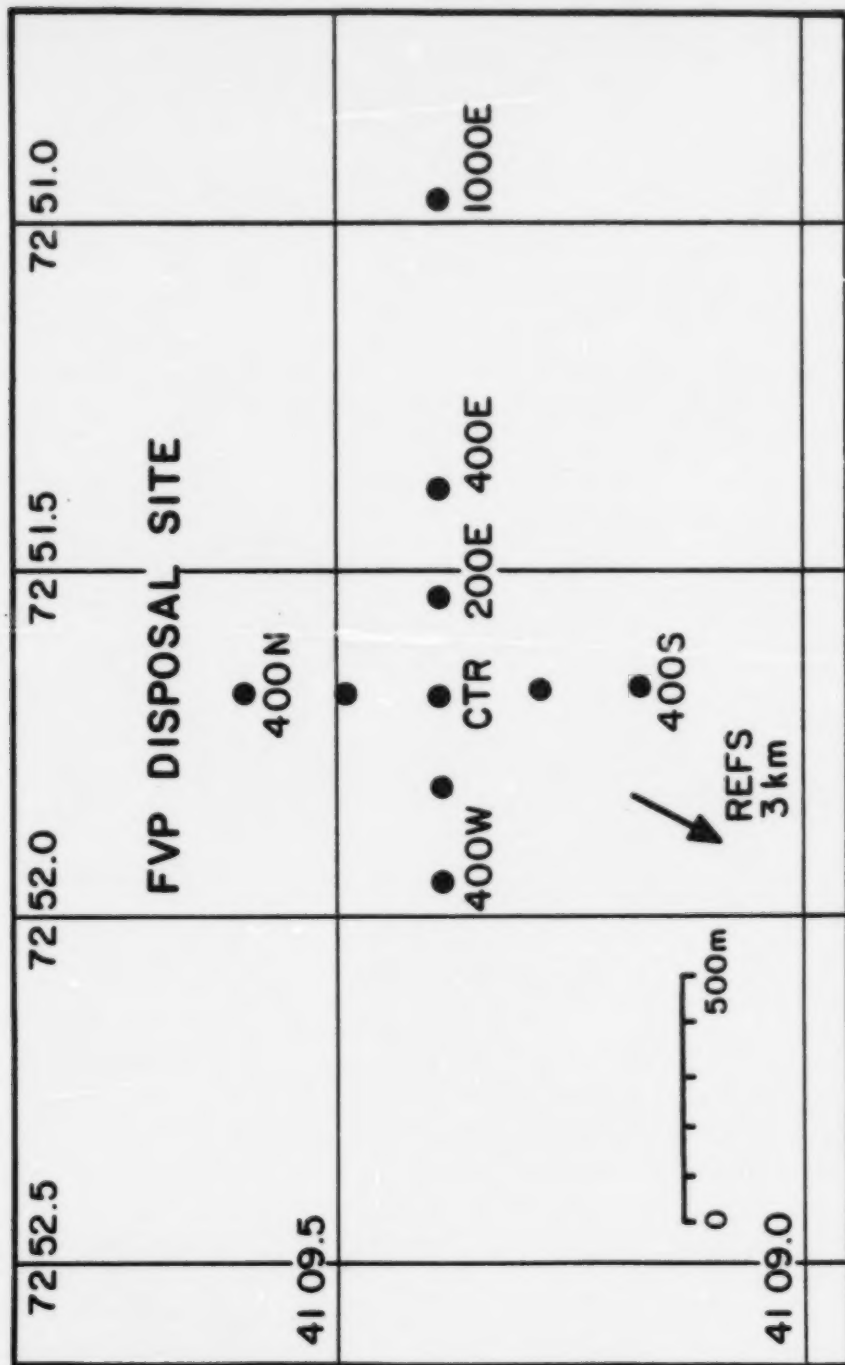


Figure 4. Placement of sampling stations in the FVP study area.

description of environmental exposure conditions. The appendices provide the raw data used in the development of this description. Appendices B through H appear in a separate volume, which is available from National Technical Information Services (NTIS).

The emphasis in subsequent phases of the exposure assessment component (reported elsewhere) was on the development and validation of internally consistent functional relationships for describing processes which control horizontal and vertical distributions of contaminants and particulate material at the disposal site. These processes are summarized in Table 2. These distributions were hypothesized to be controlled through the following chain of causal linkages:

1. wind speed and direction,
2. horizontal water motion (including those tidally induced) and vertical dispersion,
3. shear stress at the sediment-water interface,
4. flux of particulates across the sediment-water interface, and
5. partitioning (adsorption/desorption) of contaminants between dissolved and particulate phases.

Hypotheses, in the form of sub-models, are being developed for each of these linkages and will be tested using the data obtained from the laboratory and field measurements presented here.

TABLE 2. CONTROLLING PROCESSES FOR DISTRIBUTION OF
CONTAMINANTS AND PARTICULATE MATERIAL

Process	Components or compartments
Chemical Partitoning	Water Column Sediment
Biological Uptake	Water Column Infaunal Epifaunal
Sediment-Water Interactions	Physical Biological
Vertical Transport	Water Column Sediment
Horizontal Transport	Water Column Sediment

SECTION 3

DATA COLLECTION AND ANALYSIS

OVERVIEW

The physical and chemical exposure field at the FVP site in the Central Long Island Sound (CLIS) Disposal Area (Figure 5) was characterized from information collected during ERLN exposure assessment and associated cruises, deployments of in situ instruments, benthic camera surveys, and from relevant climatological and tidal data sources. Exposure assessment cruises were initiated in December 1982 to obtain baseline information concerning hydrographic parameters, water column turbidity and suspended solids concentrations, and chemical concentrations in the water column, bottom sediments and animal tissues. Following the disposal operation in May 1983, cruises continued on a monthly (through August 1983) or quarterly (December 1983 through March 1985) basis. Table 3 summarizes the ERLN exposure assessment cruise schedule. A more detailed listing of station coverage and data collection is presented in Appendix A.

Two vessels were routinely used for sampling during exposure assessment cruises to CLIS. Much of the water column work and collection of organisms, and all of the sediment sampling, in situ instrument deployment, and benthic camera surveys were conducted aboard the R/V UConn, a 65 ft research vessel operated by the University of Connecticut. The remaining water column sampling and collection of organisms were performed aboard the R/V Shang Wheeler, operated by the National Marine

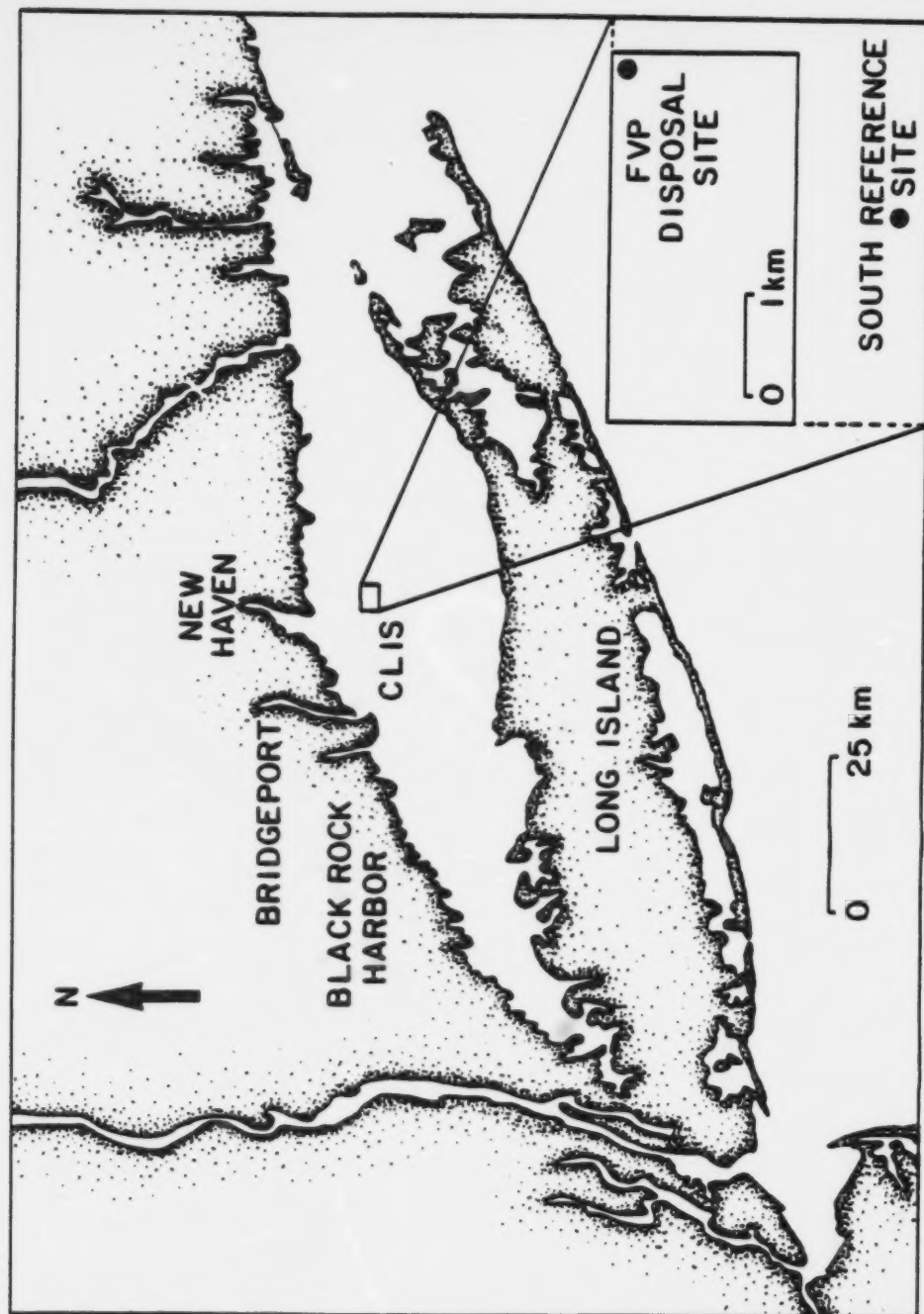


Figure 5. Long Island Sound and the location of the FVP disposal site.

TABLE 3. SUMMARY OF CRUISES IN SUPPORT OF THE EXPOSURE
ASSESSMENT COMPONENT OF THE FVP

Compartment	Date	Samples collected
Water Column	7 Apr 1983	organic chemistry
	22 Apr 1983	inorganic chemistry and physical parameters
	23 May 1983	inorganic chemistry and physical parameters
	7 Jun 1983	organic and inorganic chemistry and physical parameters
	21 Jul 1983	inorganic chemistry and physical parameters
	21 Jul 1983	organic chemistry
	31 Aug 1983	organic chemistry
	2 Sep 1983	inorganic chemistry and physical parameters
	5 Dec 1983	inorganic chemistry and physical parameters
	5 Dec 1983	organic chemistry
	21 Mar 1984	inorganic chemistry and physical parameters
	12 Apr 1984	inorganic chemistry and physical parameters
	6 Jun 1984	inorganic chemistry and physical parameters
Sediment	9 Dec 1982	bulk samples
	2 Mar 1983	bulk samples
	3 Jun 1983	bulk samples
	26 Jul 1983	cores
	1 Sep 1983	cores
	9 Dec 1983	cores
	19 Mar 1984	cores
	12 Jun 1984	cores
	11 Sep 1984	cores
	20 Dec 1984	cores
	20 Mar 1985	cores and gravity cores
(continued)		

TABLE 3 (continued).

Compartment	Date	Samples collected
Biota	17 Aug 1982	<u>Nephtys</u>
	16 Nov 1982	<u>Nephtys</u>
	16 Feb 1983	<u>Nephtys</u>
	12 Apr 1983	<u>Nephtys</u>
	22-23 Apr 1983	<u>Mytilus</u>
	24 May 1983	<u>Mytilus</u>
	2 Jun 1983	<u>Nephtys</u>
	7 Jun 1983	<u>Mytilus</u>
	3 Jul 1983	<u>Nephtys</u>
	13 Jul 1983	<u>Mytilus</u>
	10 Aug 1983	<u>Mytilus</u>
	6 Sep 1983	<u>Mytilus</u> and <u>Nephtys</u>
	18 Oct 1983	<u>Mytilus</u>
	29 Nov 1983	<u>Mytilus</u> and <u>Nephtys</u>
	20 Mar 1984	<u>Mytilus</u> and <u>Nephtys</u>
	5 Jun 1984	<u>Mytilus</u>
	12 Jun 1984	<u>Mytilus</u>
	16 Oct 1984	<u>Nephtys</u>
	17 Oct 1984	<u>Mytilus</u>

Fisheries Service in Milford, CT. Both vessels are fitted with cabins for laboratory work and have electric winches and booms to support a variety of over-the-side sampling procedures. Although buoys were generally maintained at biological effects sampling stations, characterization of small scale spatial variability dictated the use of precision ship-board navigation systems. The R/V UConn was equipped with the SAIC Navigation and Data Acquisition System, consisting of a Del Norte 540 microwave position system interfaced with a microcomputer control unit and I/O display devices (Scott *et al.*, 1984). The Del Norte triangulates ship's position based on signals sent from transponders located at Stratford Point, Stratford, CT (41° 5.467' N, 73° 3.736' W) and at Lighthouse Point, New Haven, CT (41° 8.959' N, 72° 32.553' W). This system permits station location with an accuracy of ± 5 m. SAIC's navigation system was also used in the collection of sediment from Black Rock Harbor prior to dredging (see below). The R/V Shang Wheeler is equipped with Loran-C navigational equipment, permitting station location with an accuracy of approximately ± 10 m.

DREDGING OPERATION AND COLLECTION OF BARREL MATERIAL

The source of dredged material was Black Rock Harbor, a small, highly industrialized harbor in Bridgeport, Connecticut (see Figure 5). There are a number of point and non-point discharges in and around the harbor, including a sewage treatment facility, a landfill, and industrial discharges. The dredging operation itself was designed to be a maintenance activity to remove recently sedimented material through the use of clamshell dredges. Between 10 March and 4 June 1983, approximately 302,300 cubic meters of material were removed from Black Rock Harbor (Morton *et al.*, 1984). A portion of this material (approximately 55,000 m³) was transported by barge to the FVP disposal site in CLIS between 26 April and 14 May. The remainder was used in the upland and wetland portions of the

FVP, or disposed at other locations in the CLIS Disposal Area.

In April 1982, sediment samples were collected at 37 stations in the harbor by gravity coring. These samples were analyzed for contaminant levels and physical characteristics at COE's New England Division (NED) Laboratory in Waltham, MA, and the Corps Water Quality Laboratory in Hubbardston, MA. Based upon earlier analyses, the FVP study reach was designated to include the most contaminated area (stations BR-8 through BR-32) within the harbor. This area began 400 m south of the fork in Cedar Creek and extended seaward for approximately 1,700 m, as shown in Figure 6.

Sediment to be used in subsequent laboratory biological effects studies was also collected in April 1982 from 25 sites within the study reach using a 0.1-m^2 box corer which penetrated to an approximate depth of 1.2 m. This material was transported under refrigeration (4°C) in 25 steel barrels to COE's Waterways Experiment Station (WES) in Vicksburg, MS, where it was homogenized in a clean, nitrogen purged cement truck and returned to the original barrels. Half of the barrels were sent to ERLN, and stored at 4°C until required for laboratory tests. The remainder of the barrels were kept at WES for tests in other portions of the project. A more complete description of this operation may be found in Rogerson *et al.* (1985).

Material stored in each barrel was rehomogenized prior to use in ERLN laboratory studies to ensure uniformity. The sediment was press-sieved through a 1-mm mesh stainless steel screen with a polyethylene spatula, and deposited into a round 75-gal fiberglass tank. Care was taken to ensure that all water contained in a given barrel was transferred to the fiberglass tank. The sediment was then manually stirred for approximately 45 min using paddles fabricated from commercial plastic ceiling insert grids. One- cm^2 openings in the mixing paddles allowed

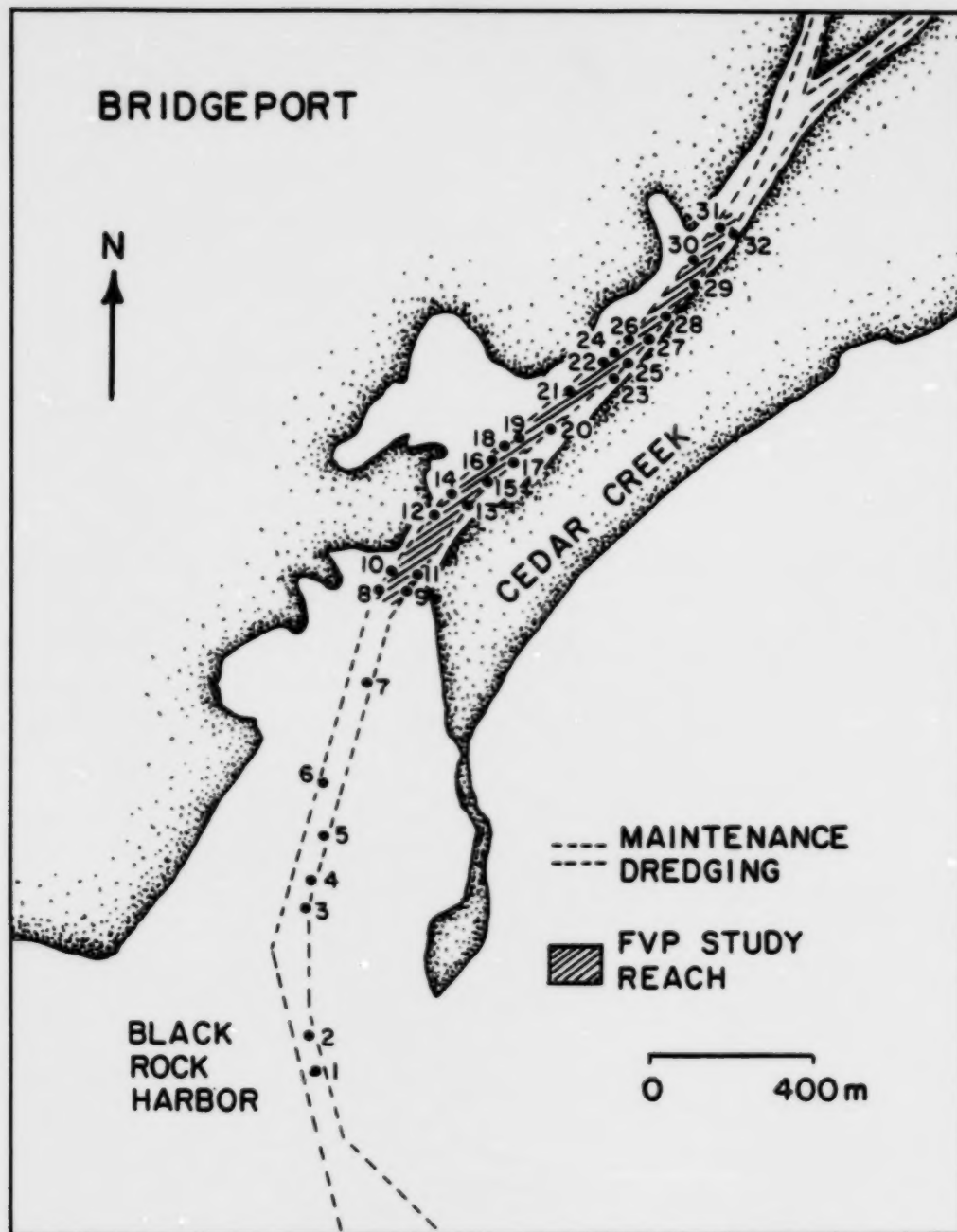


Figure 6. FVP study reach sampling stations.

the sediment to ooze through the paddle blade during the stirring process. Visual inspection of the mixture indicated that a smooth consistency had been obtained by this stirring process. The material was then distributed into 3.8-l brown glass reagent bottles. These were stored under refrigeration at 4°C. Prior to filling the 1st, 25th, and 50th bottles, approximately 400 ml of sediment were sampled for analysis of moisture content and chemical homogeneity of organic components and trace metals. Two hundred and fifty ml of each of these three splits were saved in polyethylene bottles and have been archived under refrigeration at 4°C.

FIELD PHYSICAL PARAMETERS

Physical parameters measured in the water column include vertical profiles of temperature, conductivity, salinity, dissolved oxygen (DO), and turbidity. In addition, suspended solids concentrations were measured at selected depths near bottom and at the surface for calibration of turbidity readings. Particle size analyses were carried out on selected sediment samples collected on and around the disposal mound. Sampling schemes for the water column, near-bottom, and sediment collection operations were designed to provide relevant information concerning the exposure field at the biological effects measurement stations as well as to provide insight into processes affecting the transport and fate of contaminants associated with disposed dredged material.

ERLN Exposure Assessment Cruises

Temperature, conductivity, salinity, and turbidity data were collected simultaneously through the use of a sampling cluster (Figure 7) lowered manually overboard. Sampling depth was controlled by using a metered safety line and reinforced PVC hose, which was used as part of the inorganic chemistry sampling

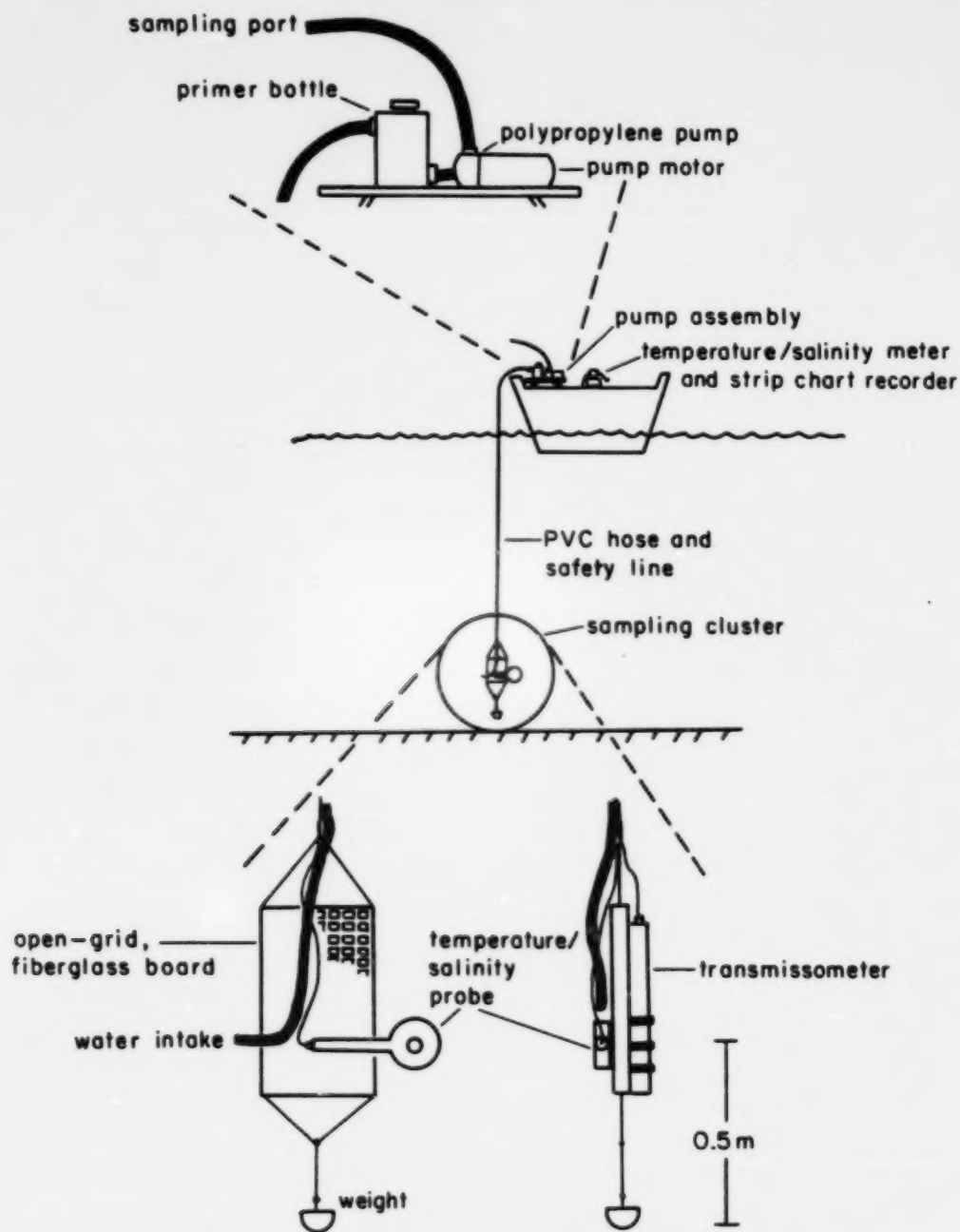


Figure 7. Physical parameters and inorganic chemistry water column sampling system.

operation described later in this section. As no attempt was made to correct for inaccuracies due to non-vertical wire angles, recorded depths were approximate values only. Vertical temperature, conductivity, and salinity profiles were obtained using a Beckman Portable model RS5-3 electrodeless induction salinometer. The resulting measurements are accurate to $\pm 0.5^{\circ}\text{C}$ for temperature, ± 0.05 millimhos/cm for conductivity, and ± 0.3 ppt for salinity. These data were entered by hand into computerized data bases. A similar procedure was used for hydrographic data collection during biological effects measurement cruises.

Profiles of light transmission were obtained using either a 10 cm or 25 cm pathlength Sea Tech., Inc. transmissometer. These transmissometers use nearly monochromatic, red-light (660 nm) LED light sources, which are relatively insensitive to absorbance by 'yellow matter' (mostly humic acids). Changes in transmission are therefore due primarily to absorbance and backscattering by suspended particulate matter. The output obtained from the transmissometer was recorded on a Hewlett-Packard electric pen strip chart recorder. The data were later transcribed by hand into computerized data bases. Percent light transmission (T) was calculated as:

$$T = (1 - T_z/T_b) \times 100$$

where T_z is transmissometer output obtained at depth z and T_b is transmissometer output obtained when the light path was completely blocked. The accuracy of transmission output from either transmissometer is $\pm 0.5\%$.

Estimates of suspended solids concentrations were obtained from selected depths in the water column (generally 1 and 3 m above the sediment-water interface [1 and 3 MAB] and 5 m below the water surface [5 MBS]) by pumping water samples through the

PVC hose and inorganic water chemistry pumping system (see Figure 7). Replicate 1 liter water samples were collected in rinsed Nalgene bottles and immediately filtered onto preweighed, 47 mm diameter Gelman GF/AE glass fiber filter pads (0.4- μ m effective pore size). The filter pads were dried for 24 h at 104°C upon return to ERLN, and reweighed in a Mettler H20T balance to obtain suspended solids weights. Sample volumes and filter weights were entered by hand into computerized data bases. In addition to providing direct measurement of suspended solids concentration, these data were used to calibrate the transmission data by regressing suspended solids concentration against percent light transmission.

Dissolved oxygen (DO) profiles were measured using a Martek Mark IV oxygen probe lowered manually overboard. Sufficient time was allowed at each sampling depth for equilibration of the probe prior to obtaining a reading. Single DO measurements taken during biological effects measurement cruises were obtained by Winkler titration of water samples collected in a Niskin bottle lowered to depth. All DO concentrations were entered by hand into computerized data bases for subsequent analysis.

Acoustic bathymetric surveys were conducted at the FVP study site to determine the thickness and extent of coverage of the disposed dredged material (Scott *et al.*, 1984). SAIC's Navigation and Data Acquisition System was used to position the R/V UCONN along a grid consisting of 33 transect lines. Each transect, spaced 25 m apart, was 800 m in length and oriented in an east-west direction. Vertical changes in dredged material thickness were detected on a point by point basis, with an accuracy of ± 20 cm. Distribution of the disposed material was further investigated through use of side scan sonar, and the REMOTS® benthic camera. The side scan sonar provided information on the aerial coverage of dredged material based upon differences in surface sediment density. See below for a

description of the REMOTS® benthic camera and the nature of data collected through its use.

Sediment grain size analyses were performed on selected samples obtained from the FVP study area for comparison with material collected in the FVP study reach prior to dredging and with that used in laboratory biological effects studies. Samples collected from the study reach were analyzed by NED. BRH barrel material was analyzed by the University of Connecticut through a combined sieve and pipet method (see Plumb, 1981). Samples were passed through a 62.5- μm screen using deionized water. The coarse fraction retained on this sieve was dried at 50°C, ground in mortar and pestle, and passed through a graded series of screens. Material collected on each screen was dried and weighed to determine the proportion of particles in 7 size classes between 62.5 and 176.8 μm . The fine fraction and that sediment from the coarse fraction passing through the 62.5- μm sieve were combined and suspended in a pipet with deionized water for successive sample withdrawal based upon Stokes settling. These samples were dried and weighed to determine the proportions in 10 size classes between 0.49 and 62.5 μm .

Surface sediments (0-10 cm deep) were collected for grain size analysis using a stainless steel Smith-MacIntyre grab sampler at biological effects stations prior to disposal on 4 March 1983, and following disposal on 3 June, 26 July, and 1 September 1983. Detailed descriptions of the methods used in obtaining and handling sediments are given in the chemical methods section below. Samples were dispersed with Calgon® (sodium carbonate and sodium tripolyphosphate) in deionized water, and analyzed using the combined sieve and pipet method. Particles were partitioned into six size classes from 0 to 62.5 μm and greater.

Bulk samples were also collected from CTR on 18 October 1983. Material from several Smith-MacIntyre grabs was homogenized by hand and subsampled for analysis at the University of California, Santa Barbara. Replicate subsamples were passed through 500 and 250- μ m sieves using filtered sea water. Particles retained on the sieves were first dried at 40°C for 24 h and then at 110°C for 30 min, and weighed to determine the proportion of the total subsample in those two size classes. The remaining material was analyzed optically using a Malvern particle sizer, which measured the proportion of particles in 16 size classes between 0 and 564 μ m. A similar analysis was performed on BRH material stored in barrels for comparison purposes.

In situ Instrument Array Deployments

An in situ instrument array was deployed at the FVP site through a Cooperative Agreement with the University of Connecticut to provide information concerning near-bottom physical processes. The Disposal Area In situ System (DAISY), developed by the University of Connecticut, is a free standing array (Figure 8) which provides near-continuous, in situ measurement of turbidity and associated hydrographic parameters, such as current speed and direction, temperature, and salinity (Bohlen, 1982; Bohlen and Winnick, 1984a). Sampling is controlled by a Sea Data model 651-8 data logger/logic control unit which collects data from three basic subsystems of instrumentation: (1) an optical array consisting of two 10-cm path length, red-light transmissometers mounted vertically at approximately 1 m above the sediment-water interface for measurement of light transmission; (2) a savonius rotor current meter (or a two-axis electromagnetic Marsh McBirney model 585 OEM current meter) with compass and vane mounted 0.5 m (1.0 m for the electromagnetic current meter) above the sediment-water interface for measurement of current speed and direction; and

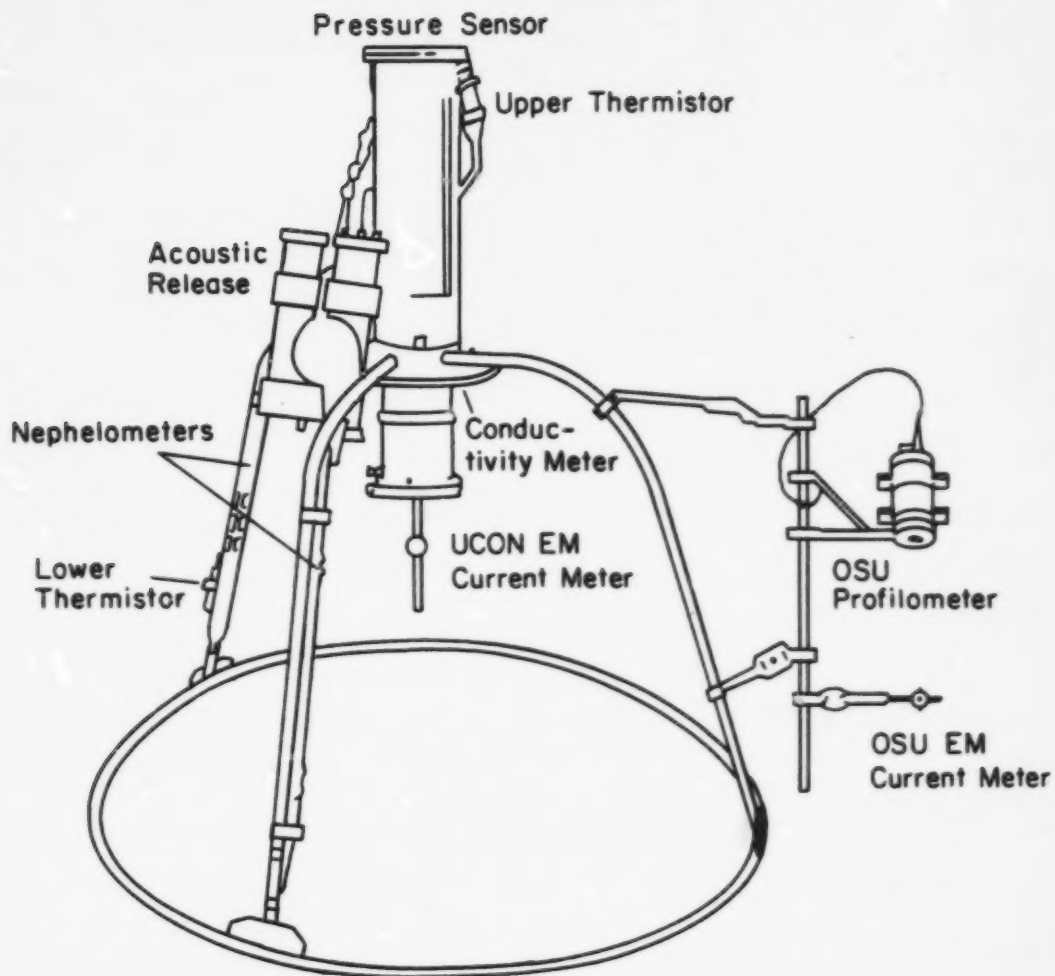


Figure 8. In situ instrumentation array developed by the University of Connecticut for measurement of near-bottom physical parameters.

(3) a mechanical water sampling system for determination of suspended solids concentrations. In addition, water temperature and conductivity measurements are obtained from sensors mounted on the frame of the array.

Temperature, salinity, current, and turbidity data were collected at 2-sec intervals during 96-sec sampling bursts which occurred four times per hour. Suspended solids samples were collected at scheduled intervals and were retrieved during maintenance operations. The data were stored on magnetic tape cassettes (data storage capacities sufficient for approximately 36 days) and were ultimately transcribed to nine-track magnetic tape for transfer to ERLN. Transmission data were calibrated to suspended solids samples collected during deployment, as well as to data obtained in laboratory studies conducted prior to and immediately subsequent to instrument deployment. The latter data were also used to correct the transmission data for instrument drift due to sedimentation and biogenic fouling of the optical windows.

With the exception of aperiodic lapses and monthly maintenance servicing, the DAISY was deployed in a near-continuous fashion from 16 August 1982 to 8 April 1983 at CTR. From 18 April 1983 to 17 August 1983, the period during which the disposal operation took place, the DAISY was deployed at 1000E. On 31 August 1983 the DAISY was moved to 400E, where it remained until 21 December 1984. An additional 3-day deployment at 400E occurred on 12-15 August 1985 to obtain near-bottom suspended solids data using an acoustic profilometer (see below). The DAISY deployment schedule is summarized in Table 4.

In addition to the basic DAISY array described above, an Edo-Western model 563, 3-MHz acoustic profilometer system developed at Ohio State University (OSU) (Bedford et al., 1982)

TABLE 4. DEPLOYMENT SCHEDULE OF THE DAISY
IN SITU INSTRUMENTATION ARRAY

Deployment site	Deployment date	Retrieval date
CTR	16 Aug 1982	3 Sep 1982
	3 Sep 1982	5 Oct 1982
	13 Oct 1982	18 Nov 1982
	18 Nov 1982	8 Dec 1982
	8 Dec 1982	27 Dec 1982
	27 Dec 1982	25 Jan 1983
	27 Jan 1983	14 Mar 1983
	14 Mar 1983	8 Apr 1983
1000E	18 Apr 1983	26 Apr 1983
	26 Apr 1983	23 May 1983
	23 May 1983	16 Jun 1983
	16 Jun 1983	29 Jun 1983
	29 Jun 1983	12 Jul 1983
	12 Jul 1983	27 Jul 1983
	27 Jul 1983	2 Aug 1983
	4 Aug 1983	17 Aug 1983
400E	31 Aug 1983	*
	25 Oct 1983	28 Nov 1983
	29 Nov 1983	16 Dec 1983
	16 Dec 1983	17 Jan 1984
	17 Jan 1984	17 Feb 1984
	17 Feb 1984	20 Mar 1984†
	20 Mar 1984	12 Apr 1984
	12 Apr 1984	14 May 1984
	14 May 1984	8 Jun 1984
	11 Jun 1984	13 Jun 1984†
	22 Jun 1984	12 Jul 1984
	12 Jul 1984	1 Aug 1984
	1 Aug 1984	17 Aug 1984
	17 Aug 1984	6 Sep 1984
	6 Sep 1984	9 Oct 1984
	9 Oct 1984	7 Nov 1984
	8 Nov 1984	27 Nov 1984
	28 Nov 1984	21 Dec 1984
	12 Aug 1985	15 Aug 1985†

* instrumentation array retrieved by commercial fisherman

† deployment with OSU acoustic profilometer

was added during deployments on 17 February and 11 June 1984, and 12-15 August 1985 to measure suspended solids concentrations over the bottom 1 m of the water column. The profilometer measured solids concentrations, based on acoustic backscatter, at 0.8-cm intervals in the water column. These data were calibrated to the suspended solids concentrations obtained from the drawn water samples described above and from diver-collected samples. Sampling control and data storage and transfer were effected in a manner similar to that described for the other DAISY subsystems. A complete description of the DAISY instrumentation array and the methods used to obtain physical parameter information may be found in Bohlen and Winnick (1984a,b, 1986), and Bedford et al. (1986a,b).

Additional Sources Of Data

Additional information concerning the physical environment of the FVP study area was obtained from a variety of sources. Surface climatological data were obtained from the National Weather Service (NOAA) station located at Sikorsky Memorial Airport at Bridgeport, CT, approximately 28 km from the disposal site. In addition to air temperature and precipitation data, these records included wind direction and speed measurements. Climatological data was also supplied by the Millstone Point Nuclear Power Plant at Millstone, CT, approximately 60 km from the disposal site. Data were transferred to ERLN on 9-track magnetic tape.

Digitized images of sea surface temperatures produced from the AVHRR sensor on the NOAA 7 satellite were available from the University of Rhode Island's Oceanographic Remote Sensing Laboratory. Images were routinely available on a twice-daily basis over the course of this study.

Data concerning fresh water runoff into Long Island Sound

was provided by the State of Connecticut Water Survey in conjunction with the United States Geological Survey for 1982, 1983, and 1984. Stream flow and suspended load data were available for the major riverine sources into CLIS. No major rivers flow into CLIS from Long Island, NY. Data were transferred by hand to computerized data files for further analysis.

Tidal height data from 5 gauging stations around Long Island Sound were obtained on 9-track magnetic tape from the National Ocean Survey (NOAA). In addition to hourly height and high-low information for several years, the results of harmonic analyses of selected periods were made available for each of the stations. A summary of the time periods covered by these records may be found in Table 5.

Additional information about sediment grain size and such bottom features as oxidation depth, surface roughness, and dredged material depth was provided by the Remote Ecological Monitoring Of The Seafloor (REMOTS®) benthic profile system developed by Marine Surveys, Inc. (Rhoads and Germano, 1982; Germano, 1983). The REMOTS® system consists of an underwater camera and optical prism mounted on a frame which is lowered to the sea bottom (Figure 9). A hydraulic piston pushes the prism into the sediment at a controlled rate to a depth of 15-20 cm. High resolution photographic images taken by the camera are analyzed with a Measurionics Corporation LMS computer image analysis system to provide estimates of modal grain size based on a grain size standard, depth of the biological mixing zone (BMD; formerly referred to as the redox potential discontinuity [RPD]); see Scott *et al.*, 1987), and information concerning other physical and biological features. A summary of the REMOTS® survey periods may be found in Table 6.

TABLE 5. SUMMARY OF TIDAL DATA OBTAINED FROM THE NATIONAL OCEAN SURVEY, NOAA

Station	High/low data	Hourly height data	Harmonic analyses
New London, CT	7 years	1979-1983	1974
Bridgeport, CT	4 years	1979-1983	1972, 1975, and 1980
Montauk, NY	8 years	1977-1983	1975, 1980
Port Jefferson, NY	7 years	1980-1983	1970-1974, and 1977
Willets Point, NY	7 years	1980-1983	1975

INTERFACE CAMERA

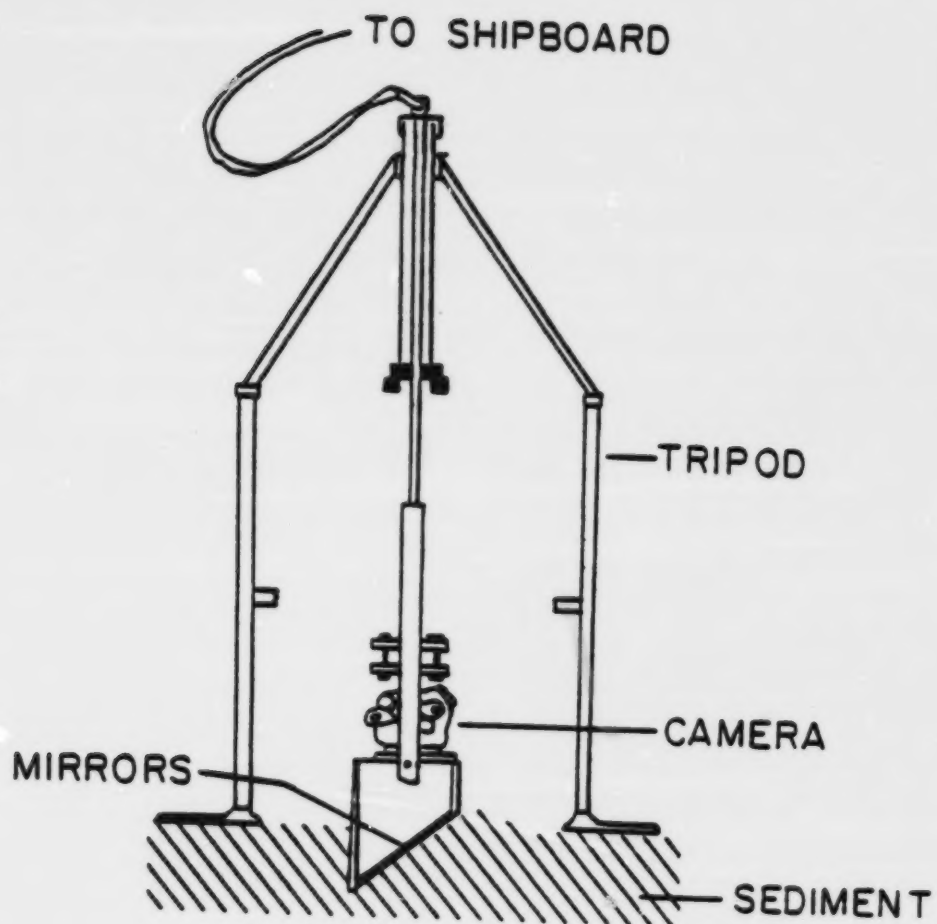


Figure 9. The REMOTS benthic camera system.

TABLE 6. SUMMARY OF REMOTS* SURVEY DATES AND STATION COVERAGE

Date	Station coverage
Aug 1982	REFS, CTR, 100N,E,S,W, 150N,E,S,W, 200N,E,S,W, 250N,E,S,W, 300N,E,S,W, 400N,E,S,W, 500E,W, 1000E, 200N/100E, 200N/100W, 200S/100E, 200S/100W, 200N/300E, 200N/300W, 200S/300E, 200S/300W, 200N/500E, 200N/500W, 200S/500E, 200S/500W, 400N/100E, 400N/100W, 400S/100E, 400S/100W, 400N/300E, 400N/300W, 400S/300E, 400S/300W, 400N/500E, 400N/500W, 400S/500E, 400S/500W
Mar 1983	REFS, CTR, 100S,W, 150N,E,W, 200N, 250E,S, 300S, 400N,S, 500E,W, 1000E
May 1983	reconnaissance, same as Aug 1982
May 1983	CTR, 100S,W, 150N,E,W, 200N, 250E,S,W, 300N,S, 400N,E,S, 500E,W, 1000E,W, 1500E
Jun 1983	REFS, CTR, 100W, 150E, 200N,S, 250N,E,W, 300E,S, 400N,S, 500E,W, 750E,W, 1000E,W, 1100E,W, 1200E,W, 1300E,W, 1400E,W, 1500E,W, 200N/300E, 200N/300W, 200S/300E, 200S/300W
Jul 1983	REFS, CTR, 100W, 150E, 200N,S, 250N,E,W, 300S, 400N,S, 500E,W, 750W, 1000E,W, 200N/300E, 200N/300W, 200S/300E, 200S/300W
Aug 1983	same as Jul 1983
Dec 1983	same as Jul 1983
Mar 1984	same as Jul 1983
Jun 1984	REFS, CTR, 100W, 150E, 200N,S, 250N,E,W, 300S, 400N,S, 500E,W, 750E,W, 1000E,W, 200N/300E, 200N/300W, 200S/300E, 200S/300W
Sep 1984	same as Jun 1984
Dec 1984	same as Jun 1984
Mar 1985	same as Jun 1984
Jun 1985	same as Jun 1984

FIELD CHEMICAL DATA

Trace metals and organic compounds were measured in the water column, sediments, and the tissues of deployed Mytilus edulis and native Nephtys incisa. Due to differences in analytical requirements and contamination sources between inorganic and organic water column determinations, separate sample collections were made for each. Common samples were collected of sediments and organisms. Sampling schemes for the water column, sediments and organisms were designed to provide relevant information concerning the chemical exposure field at the biological effects measurement stations as well as to provide insight into processes affecting the transport and fate of contaminants associated with dredged material. A detailed listing of station coverage is given in Appendix A.

Water Column Collection Methods

Inorganic Samples--

Instrumentation for trace metal water collection and physical measurements was bundled into a 30-m array of two cables, hose, and polypropylene rope (Figure 7). A polyethylene covered lead weight (ca. 10 kg) was attached 0.5 m below the hose intake and instrument array. A description of this instrument array is given above. The hose, rope, and cable bundle were marked in one-meter graduations. The array was lowered from the deck of the ship until it rested on the bottom, and then raised 0.5 m to begin sampling 1 m above bottom.

Seawater samples for inorganic analysis were obtained through a 30-m by 16-mm (internal diameter) nylon reinforced, clear PVC hose. The intake end of the hose was equipped with a polyethylene/PVC check valve. The pump was a LFE Eastern model MD-30 capable of delivering 30 l/min. Actual delivery to deck was about 20 l/m. The pump head casing was made of glass-filled

polypropylene with a magnetically coupled impeller encased in polypropylene. The teflon impeller bearing rotated on a stationary ceramic spindle. Sea water did not contact any metallic motor parts prior to or during sample collection. A description of this type of system has been given by Kingston et al. (1983). A four-liter polyethylene prime bottle served to provide a steady head of water from the hose to the pump.

Hose, prime reservoir, and pump were well flushed for approximately two min before samples were taken at a given depth. All water samples were collected in duplicate (or triplicate) so that sample reproducibility could be ascertained. One-liter samples were filtered with prewashed and tared Gelman glass fiber filters (0.4- μ m effective pore size) to determine the suspended solids concentration at each depth, as described above.

A separate set of seawater samples was collected at each location and depth for the determination of trace metal concentrations. These water samples consisted of whole water, particulate, and dissolved fractions. The amount of time needed to collect a complete set of samples at a given depth was approximately 2 minutes; therefore, a single water mass should have been sampled. Whole water samples were collected by filling precleaned and tared polyethylene bottles to approximately 250 ml. Particulate and dissolved water samples were obtained by filtering approximately 250 ml of sea water through a precleaned and tared Nuclepore filter (0.4- μ m pore size, 47-mm diameter, polycarbonate membrane). The filtration was accomplished with an all-plastic inline nitrogen pressure apparatus (Figure 10) similar to that described by Hanson (1981). Each filter was stored separately in an acid stripped, polystyrene petri dish. Filtrate was collected for the dissolved sample in a precleaned and tared 250-ml polyethylene bottle.

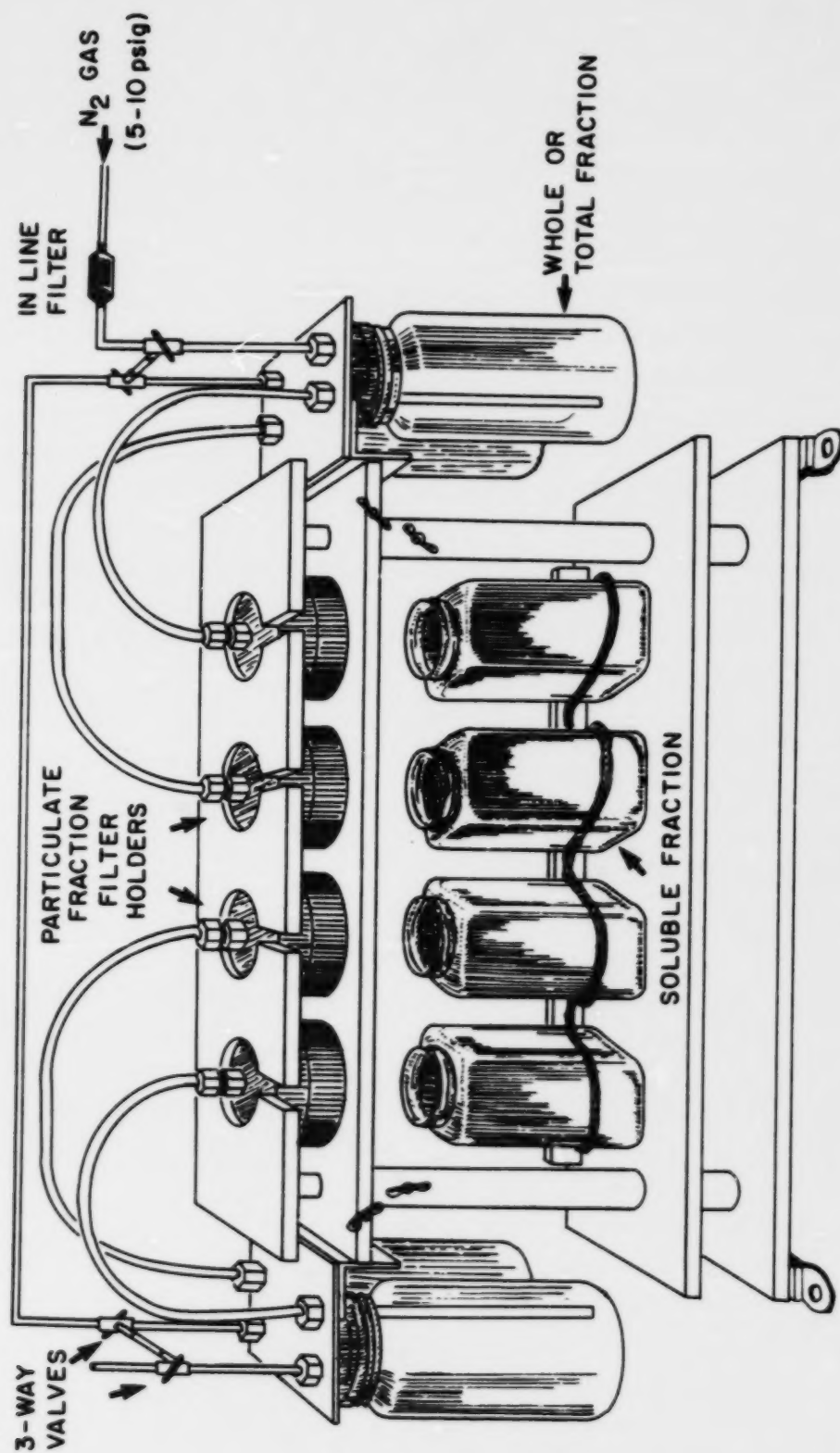


Figure 10. Filtration system used in inorganic water column sampling.

Seawater samples cannot be stored for long periods of time if both particulate and dissolved fractions are to be analyzed (Kremling, 1983). In coastal or estuarine waters with high suspended solids levels, bacterial action can alter the equilibrium between the particulate and dissolved phases in stored sea water samples. All samples were therefore filtered onboard within 1 hour of collection.

Seawater sample sizes collected for whole water, particulate, and dissolved fractions were determined within 10 hours of collection. Sample bottles were reweighed, and whole water and dissolved samples were acidified with 250 μ l of Ultrex concentrated HNO_3 to a pH of approximately 2. This was necessary to minimize adsorption of trace metals to the sample container walls (Masse et al., 1981).

Organic Samples--

Water column samples were obtained for organic analysis from 1 MAB and 5 MBS at the biological effects measurement stations. Separate samples were collected for whole water, and for dissolved and particulate fractions. All water samples were collected using a teflon hose with an outside stainless steel braid and stainless steel fittings attached to a Jupiter teflon centrifugal pump (Fluorocarbon Process Systems). The pump was floated in a fiberglass box at the sea surface and the hose was attached to a weighted line and deployed at the desired depth (Figure 11).

Whole water--Whole water samples were pumped from the desired depth and collected in 3.8-l brown glass bottles. These were empty solvent bottles that were rinsed twice with methylene chloride just prior to the cruise, and rinsed with seawater immediately before the sample was taken. Generally, 6 bottles were filled for each sample for a total volume of 22.7 l.

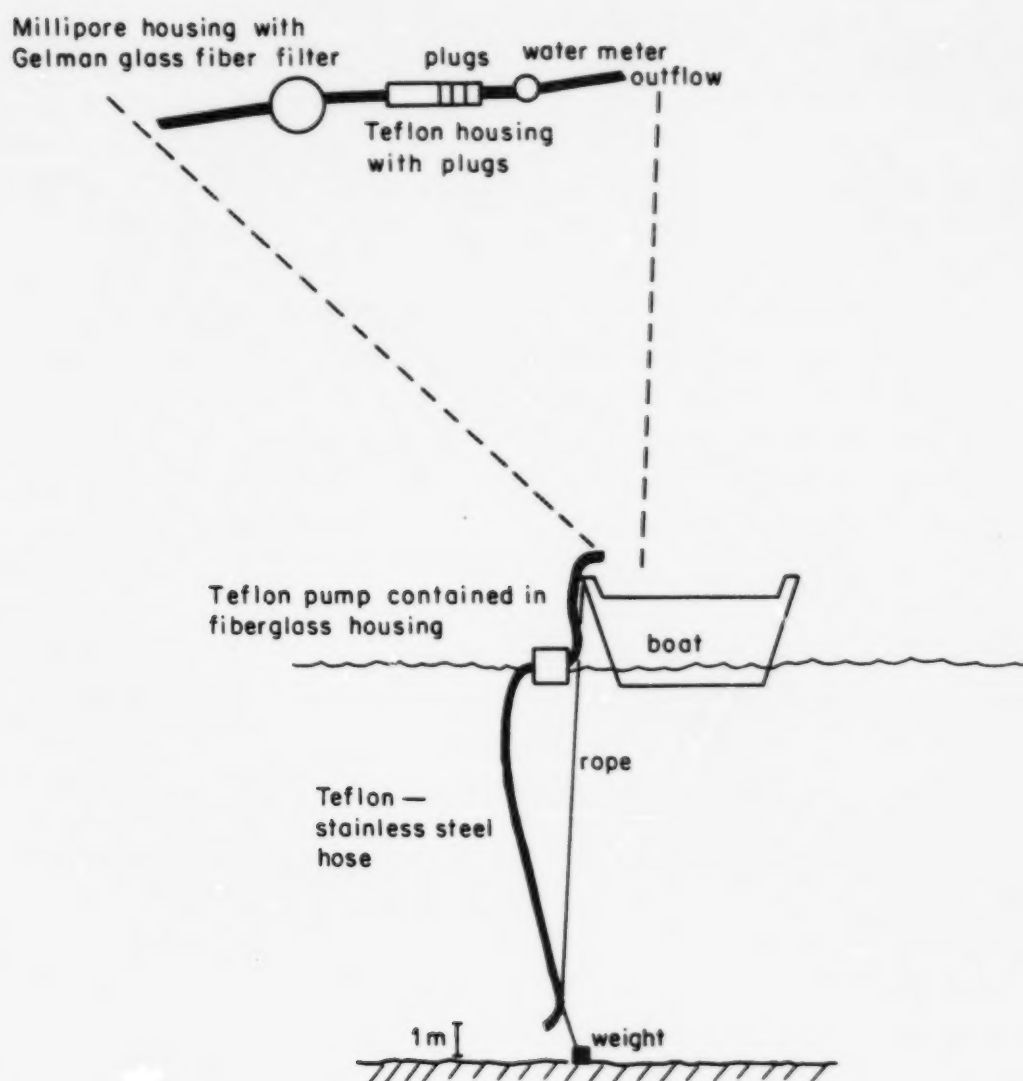


Figure 11. Organic chemistry water column sampling system.

One hundred and fifty ml of methylene chloride was added to each bottle and the sample was capped and shaken for a few minutes. The bottles were returned to the laboratory and refrigerated overnight. All sample extractions (see below) were completed the day following collection. Field blanks for whole water samples were taken by opening a sample bottle on the deck of the ship for the amount of time normally required to collect a sample. Methylene chloride (150 ml) was added and the bottle was capped and stored as above.

Dissolved and particulate fractions--Seawater pumped from the desired depth was passed through a Gelman A/E glass fiber filter (0.3 μ m, 29.3 cm diameter) held in a Millipore stainless steel holder and then through a teflon housing containing 3 polyurethane foam plugs (Identi-plug, size c, VWR Scientific). Prior to use, the filters were wrapped individually in aluminum foil and muffled at 450°C for 6 hours. The foam plugs were placed in the housing and cleaned with two 50-ml washes of acetone followed by two 50-ml washes of hexane. This step was repeated, with the second wash collected as a blank (called a laboratory blank). Plugs were then flushed with two 50-ml washes of acetone and 200 ml of deionized water after which the housings were capped with brass or stainless steel endcaps for transport to the field.

Just before sampling, a muffled filter was removed from the aluminum foil using metal forceps and placed into the filter housing. A precleaned teflon housing containing the foam plugs was placed in line behind the filter. A Recordall Model 15 water meter, used to measure the total volume of water passing through the system, was placed downstream of the filter and foam plugs (Figure 11). During sampling, the flow rate was periodically measured (using a graduated cylinder and stopwatch) and maintained at approximately 2,000 ml/min. Filter back pressure was monitored and maintained at about 10 psi. Generally,

approximately 100 l were passed through the system for each sample.

After each sample was collected, the filter was removed from the holder using metal forceps, folded, placed into a precleaned stainless steel bottle, and frozen until analyzed. The foam plug housing was recapped and wrapped in aluminum foil for return to the laboratory. These were stored in the refrigerator overnight, and the plugs were extracted the following day.

Filter field blanks were taken by briefly placing a clean filter into the filter housing. The filter was then removed and put into a precleaned stainless steel bottle for return to the laboratory, along with the field samples. Foam plug field blanks were taken by uncapping a plug housing containing plugs on the deck of the ship and leaving it open for the approximate amount of time required to connect plug housings to the sampling gear. These samples were then handled in a manner identical to the other foam plug samples.

Sediment Collection and Storage

Replicate sediment samples were collected from separate 0.1-m² Smith-MacIntyre grab samples at each station. Initially (see Table 3), samples were scraped from the surface of the grab and stored in acid-cleaned polyethylene bottles for inorganic determinations (250 ml) and soft glass bottles for organic determinations (500 ml). Starting in July 1983, cores were taken with acid-cleaned acrylic plastic tubes (15 cm length, 7.5 cm internal diameter) from the Smith-MacIntyre grabs. All core liners and end caps were labeled with a unique sample number. Following deployment and retrieval of the grab sampler, the top trap doors were opened and the sampling tubes were carefully inserted into the collected sediment. Four sampling cores were

inserted into each grab sample by using hand pressure only. An end cap was placed onto the top of each core. Core samples were removed by carefully inserting a plastic-gloved hand into the sediment next to the core and sealing the bottom of the core with the palm of the gloved hand. The core sample was then pulled up and out of the sediment and fitted with a bottom cap, carefully rinsed with seawater, and stored in an insulated box containing dry ice. Samples were transferred to a freezer (maintained at -20°C) within 10 hours of collection.

Organism Collection Methods

Mytilus--

Deployment and retrieval scheme--Mussel samples used for CLIS transplant exposures were collected near the mouth of Narragansett Bay, RI ($41^{\circ} 29.40' \text{ N}$, $71^{\circ} 24.0' \text{ W}$). These mussels were characterized biologically and chemically at the time of each collection. The Narragansett reference population served as a convenient source of relatively uncontaminated mussels. In addition, chemical and biological parameters measured in these animals could be used to determine if the act of transplantation was detrimental to the organisms.

The configuration of the transplant apparatus at each station was that previously tested for the Coastal Environmental Assessment Studies (CEAS) program in Narragansett Bay (Phelps and Galloway, 1980). It employs diver-serviced caged mussels suspended approximately 1 m above the sediment surface. Station location was marked by a permanently moored spar buoy (Figure 12), constructed of high impact ABS plastic filled with polyurethane foam and weighted by a plastic encased concrete weight attached by galvanized pipe. It was painted international orange with a white reflective horizontal stripe as dictated by the United States Coast Guard (USCG) for Special Purpose buoys. The buoy was moored to a 150-kg concrete block by 1-cm cable

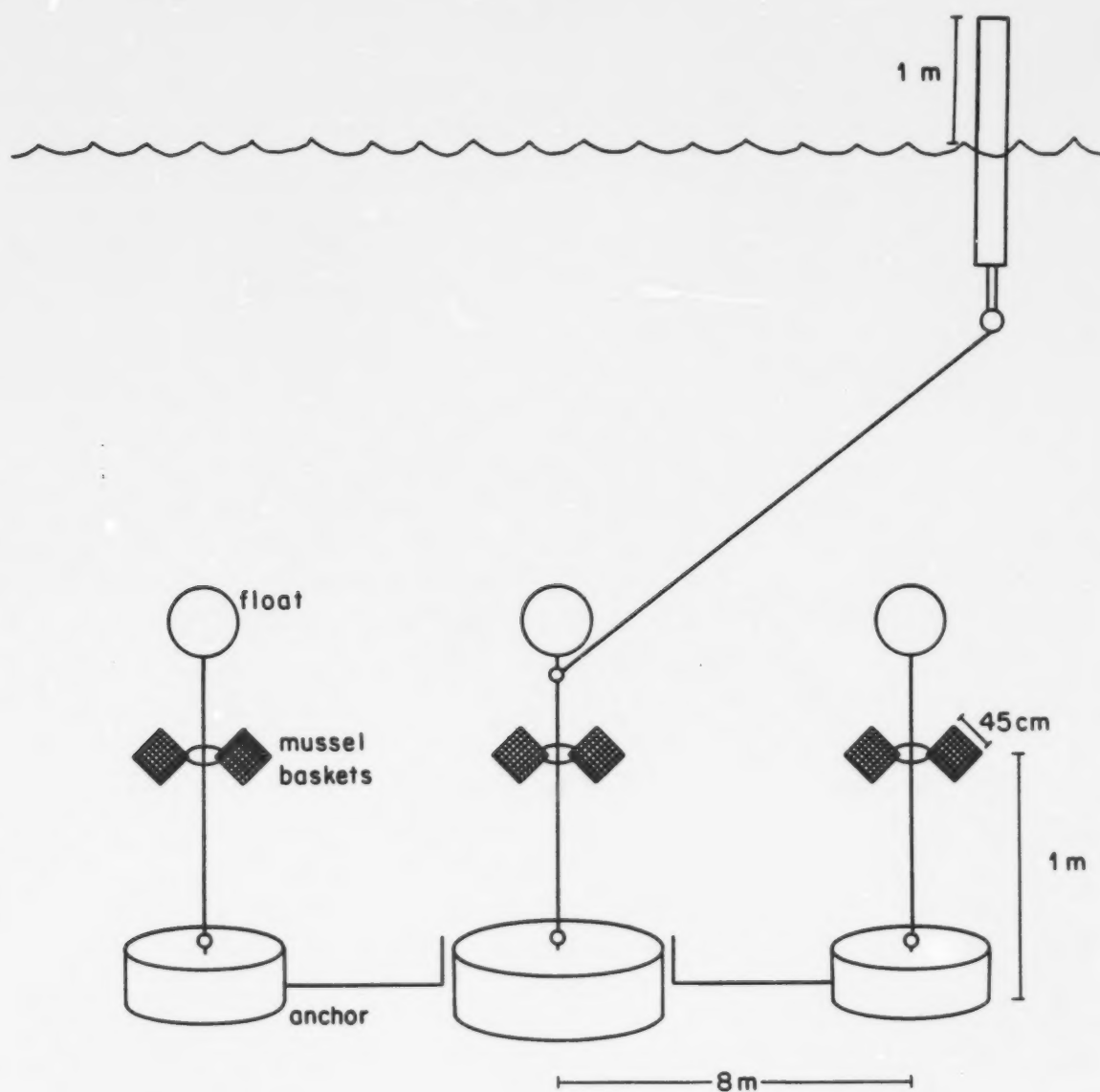


Figure 12. Caged mussel mooring system.

encased in polypropylene.

Satellite stations were used to deploy the caged mussels. Each satellite consisted of two or more 20-cm hard plastic floats tied with polypropylene line to a small concrete mooring weight (Figure 12). One hundred mussels were placed in polypropylene test baskets with snap-on lids (Mallinkrodt), which were suspended from loops in the polypropylene line approximately 1 m above the sediment surface. A nylon line, staked in proximity to the main mooring anchor, was run to each satellite station. Divers descended on the main mooring anchor line, located the stakes and followed the nylon line to the satellite stations to deploy and retrieve cages of mussels. Satellite deployments were used to assure station integrity in the event that the surface buoy and the main mooring anchor were destroyed or moved. An acoustic pinger was placed on the mooring weight of each satellite station for additional aid in relocation.

Mussels were generally deployed for 1-month periods at each of the biological effects measurement stations (see Table 3 for retrieval schedule). Mussels were retrieved from single baskets at each station and packaged in labeled polyethylene Zip-Loc bags (inorganic analysis) or wrapped in aluminum foil (organic analysis). Because all of the mussels retrieved from a given station on each sampling date were obtained from a single basket, sampling error could not be determined (see Hurlbert (1984) for a discussion of this type of replication problem). Retrieved mussels were frozen within 10 hours of collection, and stored at -20°C until analyzed.

Nephtys--

Collection methods--All Nephtys incisa were collected with a 0.1-m² Smith-MacIntyre grab sampler. Worms were separated from the sediment by gently wet-sieving the sediment through a

stainless steel screen (0.5 mm). Individual worms were hand-picked from the screen with forceps and placed into precleaned, tared 6-ml vials (inorganic analysis) or into soft glass containers (organic analysis). Samples were frozen after collection and stored at -20°C until processed for chemical analysis.

Sample Preparation and Analysis

Inorganic Samples--

Figure 13 shows an overall flow diagram of the trace metal sample preparation and analysis scheme used in this study. Data generated during the preparation of samples (e.g. dry weight, wet weight, volume, etc.) were entered into a computerized data base for use in subsequent analyses (see Section 4 for a description of data bases and data management procedures).

Seawater sample preparation--All tasks involving the preparation of seawater samples for trace metal analysis, including container cleaning and reagent preparation, were performed in a class 100 clean room. This clean room was specially modified to be free of exposed metal surfaces (Moody, 1982; Murphy, 1976; Patterson and Settle, 1976). Containers used in collection, storage, and coprecipitation were soaked for 48 hr in heated (53°C) 2N HNO_3 , for 24 hr in heated deionized water, and rinsed in heated deionized water (Laxen and Harrison, 1981). The water bath used in this procedure was made of polyethylene, while the heating elements and thermostat sensor were teflon jacketed.

Following shipboard separation of the seawater particulate fraction (see above), filter samples were dried on a laminar flow, class 100 clean bench and weighed. Filters were placed into 6-ml polyethylene vials with 5 ml of 2N HNO_3 producing an effective concentration factor of 50. Dissolution was enhanced

INORGANIC ANALYSIS FLOW CHART

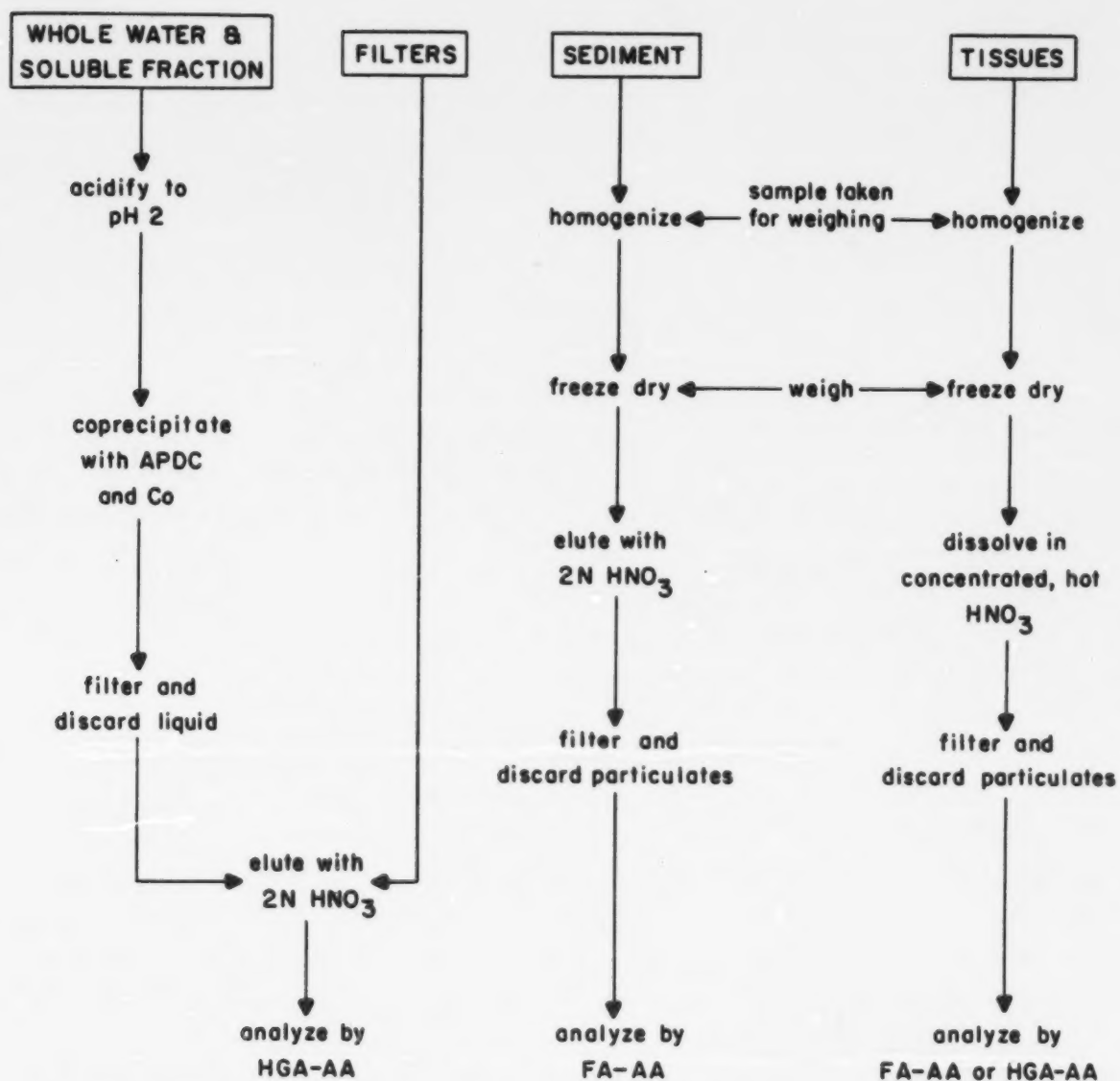


Figure 13. Inorganic chemical analysis flow chart.

by maintaining the sample vials in a sonicated water bath for a minimum of 24 hr. These samples were stored for later analysis.

No chemical separation was used to prepare Fe, Mn, and Zn in whole water samples prior to analysis. One-ml subsamples (3 to 5 replicates of each sample) were pipetted into 1.2-ml polyethylene vials (acid cleaned) and 100 μ l of concentrated Ultrex HNO_3 was added. The vials were fitted with snap caps and stored for later analysis.

Cu, Cd, Pb, and Ni were separated and concentrated from samples of both whole water and dissolved fractions with a modified coprecipitation technique first described by Boyle and Edmond (1975). After weighing and acidifying the 250-ml whole water or dissolved samples, 1 ml of CoCl_2 solution (approximately 340 ppm Co) was added to the sample and shaken well. Three ml of 2% ammonium pyrrolidine dithiocarbamate (APDC) was added, and the sample was again shaken well. A yellowish-green precipitate formed immediately. Samples were allowed to sit for 0.5 to 1.5 hr and vacuum-filtered through acid cleaned, 47-mm diameter, 0.4- μ m pore size, polycarbonate membrane filters (Nuclepore). Bottle and funnel rinses and filter washes were performed with approximately 50 ml of deionized water. Filters and precipitate were placed in 6-ml polyethylene vials with 5 ml of 2N HNO_3 . Samples were sonicated overnight to speed and ensure dissolution of precipitate. Subsamples were taken for elemental determination and for measurement of Co recovery. This preparation method yielded an approximate 50:1 concentration of analytes with recovery efficiencies of 92-98%.

The Co solution used in the coprecipitation reaction was routinely cleaned of trace metal impurities by adsorbing Co ions on Dowex 1-48 resin as described by Kraus and Moore (1953) and Huizenga (1982). Co was separated from other trace metal

contaminants by controlling the HCl concentration during the elution step. The APDC water solution used in the coprecipitation reaction was cleaned of trace metal impurities by triple extraction with methyl isobutyl ketone (MIBK) as opposed to carbon tetrachloride as used by Boyle and Edmond (1975). The APDC solution was prepared in 100-ml quantities and stored under refrigeration at 3°C. The APDC solution was found to be stable for at least one month if kept in the dark under refrigeration.

Sediment sample preparation--Samples collected during early sediment cruises (i.e. those collected by scraping the top of a grab sample) were thawed and then homogenized by stirring with a stainless steel spatula. Core samples taken on all subsequent sediment cruises were partially thawed, sectioned, homogenized, and then subsampled for analysis. Each core sample (previously stored at -20°C) was held in a refrigerator maintained at 3°C for 8 to 9 hours prior to sectioning. This technique of slowly warming the core samples allowed sectioning to be done without complete thawing. Core samples were placed on a piece of aluminum foil (muffled at 450°C) and pushed (2 cm at a time) out of the core liner. Each 2-cm increment was cut with a stainless steel spatula and transferred to an acid cleaned and muffled (450°C) glass container. To avoid cross contamination of core sections during the cutting process the spatula was washed with deionized water after each cut. At least 5 sections (2-cm length) could be cut from each core. A sixth section of varying thickness was obtained from several of the cores.

Following sectioning, sediment samples were thawed completely and homogenized by stirring with a spatula. Aliquots of wet sediment (approximately 5 g) were transferred to tared, 60-ml acid cleaned, polyethylene bottles. Replicate aliquots of several core sections were taken to determine the precision of

the methodology used to cut and mix sediment. These data are presented in Table 7. Wet weights of all samples were then determined. Sample aliquots were frozen again and freeze-dried in a Virtis lyophilizer (Model No. 10-145MR-BA) for 2 days. Dry weight of each sample was then determined. Dried sediment samples were acidified with 50 ml of 2N HNO_3 (reagent grade), sealed with a polyethylene screw cap, and stored at room temperature for two days. Samples were vented daily to prevent rupturing of the plastic containers and mildly shaken to resuspend the sediment. Samples were gravity filtered through acid washed (2N HNO_3) Whatman 42 filter paper into 60 ml, acid cleaned, polyethylene bottles so that insoluble residue would not interfere with subsequent atomic absorption analysis. A measured portion of each sample (1 to 5 ml) was treated with 30% H_2O_2 to oxidize soluble, organic, surface-active components which passed through the filter. Generally, 1 ml of peroxide was added to each ml of subsample taken for analysis. This oxidation step was necessary if the sample was to be analyzed by heated graphite atomization (HGA) atomic absorption (AA) as described below.

Mytilus sample preparation--Mussel samples were analyzed as composites of 5 individual mussels selected at random from retrieved animals. Partially thawed mussels were shucked using stainless steel instruments. As a precaution against cross-contamination, instruments used in shucking were cleaned between samples by rinsing with deionized water. Mussel meat and frozen shell liquor were collected in tared, acid cleaned, 250-ml pyrex beakers and covered with pyrex watch glasses. Wet weights were recorded and samples were dried to constant weight at 95°C. Samples were stored in a dessicator until attaining room temperature, at which time dry weights were determined.

Dry mussel samples were decomposed with 100 ml of concentrated HNO_3 (reagent grade) added in 20-ml aliquots over

TABLE 7. PRECISION OF TRACE METAL SEDIMENT ANALYSES

Metal	Average coefficient of variation	
	CTR	REFS
Fe	18.5 (12)*	0.6 (17)
Mn	6.2 (12)	2.0 (17)
Zn	12.0 (12)	0.8 (17)
Cu	7.5 (12)	1.0 (17)
Pb	4.1 (12)	0.6 (16)
Cd	3.9 (12)	4.8 (17)
Cr	30.0 (12)	0.5 (17)
Ni	11.4 (12)	3.4 (17)

* Number of sediment samples used to calculate the average percent. Each sample composed of 2 or 3 replicates.

a period of 2 days. Samples were allowed to cold digest during the period of acid addition. Mussel tissue was well broken down before gentle heat was applied. Samples were then slowly heated to just below their boiling point (approximately 60°C) and maintained at that temperature for 8 hr. This process was repeated over the course of several days, after which time samples were evaporated to incipient dryness. Residue from each sample was reconstituted with 20 ml of 2N HNO₃ and filtered through acid washed Whatman 42 filter paper into a 50-ml volumetric flask. The sample beaker was rinsed with several 10 to 15-ml washes of 5% HNO₃ which were also filtered and combined with the initial solution. The final sample solution was brought to 50 ml by addition of 2N HNO₃. Sample solutions were poured into 60-ml acid cleaned, polyethylene bottles and stored for later analysis. A 5-ml aliquot of the sample solution was oxidized with 30% hydrogen peroxide to destroy surface-active soluble organic compounds that passed through the filter.

Nephtys sample preparation--Individual Nephtys samples contained in tared labeled vials were weighed and then freeze dried in a Virtis lyophilizer until constant weight was obtained (approximately 2 days). One ml of concentrated nitric acid (reagent grade) was added to each vial to decompose dried worm tissue. Samples were allowed to cold digest for 24 hours and inserted into a thermostatically controlled (at 50°C) graphite heating block and heated for several days. During the heating period an additional 1 ml of nitric acid was added to the samples. Two ml of 30% hydrogen peroxide was added after the volume of acid was reduced to approximately 0.2 ml. The peroxide was added slowly in 100- μ l aliquots until the first ml was added. The second ml of peroxide was added after the oxidation reaction had subsided. Samples were again reduced to near dryness and allowed to cool. Five ml of 2N HNO₃ was added to each sample vial to reconstitute precipitated salts. Vials were

capped and stored for later analysis.

Atomic absorption analysis--Flame atomization (FA) atomic absorption (AA) was conducted with a Perkin-Elmer (Model 5000) AA spectrophotometer. HGA-AA determinations were conducted with Perkin-Elmer Model 500 or 2100 HGA units coupled to Perkin-Elmer Model 5000 or 603 AA instruments, respectively. The model 5000 AA was retrofitted with a Zeeman HGA background correction unit and the model 603 was equipped with a D2 arc background correction system. Both HGA units were equipped with auto injectors.

FA-AA and transient HGA-AA signals were recorded with Perkin-Elmer strip chart recorders (Model 56) and sent directly to Perkin-Elmer Model 3600 data station microcomputers. Transient signals were reduced to peak height and areas for each element determined. Standard curves were used to determine concentrations in unknown samples. Polynomial regression algorithms used to calculate standard curves are described by Rugg and Feldman (1980).

Seawater elemental determinations made with chemical separation procedures described previously were corrected for chemical yield (i.e. Co recovery) and calculated in units of $\mu\text{g/kg}$ by using the weight of seawater determined for each seawater sample. Seawater samples determined by direct injection did not need to be corrected for weight/volume of seawater collected.

Instrument setup conditions for all FA-AA and HGA-AA determinations are shown in Table 8, and are similar to those described in "Methods for Chemical Analysis of Water and Wastes" (U.S. EPA, 1979) and those found in manufacturer's reference manuals. AA instruments were calibrated each time samples were analyzed for a given element. Instrument calibrations were

TABLE 8. INSTRUMENT OPERATION PARAMETERS FOR TRACE METALS ANALYSIS

Parameter	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Whole Water and Dissolved								
AA model	PE 603	Z/5000	PE 603	Z/5000	Z/5000	Z/5000	PE 603	Z/5000
Wavelength nm	228.8	357.9	324.7	248.3	279.5	232.0	283.3	213.9
Bandwidth nm	0.7	0.7	0.7	0.2	0.2	0.2	0.7	0.7
Light source	EDL	HC	HC	HC	HC	HC	EDL	EDL
Watts/Current	5w	20ma	15ma	30ma	20ma	25ma	10w	4.5w
Bkgd correction	none	Zeeman	none	Zeeman	Zeeman	Zeeman	none	Zeeman
Time pk ht/area	4 ht	5 area	4 ht	5 ht	5 ht	5 ht	4 ht	11 ht
Tube/boat	plain	boat	plain	plain	plain	pyro	plain	plain
Inj vol μ l	10	5	10	5	10	10	20	5
Matrix*	(1)	(5)	(1)	(2)	(2)	(3)	(1)	(2)
Matrix mod μ l	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Dry °C	100	200	100	110	110	110	100	110
Ramp sec	10	5	10	5	5	5	10	10
Hold sec	10	25	20	20	20	20	20	20
Ash °C	450	1400	1000	1000	1000	1000	500	450
Ramp sec	10	5	10	5	5	5	10	5
Hold sec	10	10	10	10	10	20	10	10
Atomize °C	2000	2500	2500	2500	2400	2400	2150	1500
Ramp sec	0	0	0	1	1	0	0	7
Hold sec	5	4	5	3	3	4	5	4
Ar flow cc/min	50	0	0	50	50	50	0	50
Burn out °C	2600	2600	2600	2600	2600	2600	2600	2600
Ramp sec	1	1	1	1	1	1	1	1
Hold sec	3	3	3	3	3	3	3	3
Total sec/inj.	49	52	59	48	48	58	59	150**

(continued)

TABLE 8 (continued).

Parameter	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Particulate								
AA model	Z/5000	Z/5000	Z/5000	Z/5000	Z/5000	Z/5000	Z/5000	Z/5000
Wavelength nm	228.8	357.9	324.7	248.3	279.5	232.0	283.3	213.9
Bandwidth nm	0.7	0.7	0.7	0.2	0.2	0.2	0.7	0.7
Light source	EDL	HC	HC	HC	HC	HC	EDL	EDL
Watts/Current	4.5w	20ma	15ma	30ma	20ma	25ma	7.5w	4.5w
Bkgd correction	Zeeman	Zeeman	Zeeman	none	none	Zeeman	Zeeman	Zeeman
pk ht/area	5 ht	5 area	5 area	2 ht	2 ht	5 ht	5 ht	7 ht
Tube/boat	boat	boat	boat	fl/cup	fl/cup	pyro	boat	boat
Inj vol μ L	10	5	20	100	100	10	10	5
Matrix	(4)	(5)	(5)	(5)	(5)	(3)	(6)	(6)
Matrix mod μ L	10	n/a	n/a	n/a	n/a	n/a	5	5
Dry °C	200	200	200	n/a	n/a	110	200	200
Ramp sec	5	5	10	n/a	n/a	5	5	5
Hold sec	25	25	35	n/a	n/a	20	25	25
Ash °C	900	1400	1200	n/a	n/a	1000	600	450
Ramp sec	5	5	5	n/a	n/a	5	10	10
Hold sec	20	10	10	n/a	n/a	20	10	20
Atomize °C	1900	2500	2300	n/a	n/a	2400	2300	1500
Ramp sec	0	0	0	n/a	n/a	0	0	0
Hold sec	4	4	4	n/a	n/a	4	4	6
Ar flow cc/min	20	0	50	n/a	n/a	500	20	50
Burn out °C	2600	2600	2600	n/a	n/a	2600	2600	2600
Ramp sec	1	1	1	n/a	n/a	1	1	1
Hold sec	3	3	3	n/a	n/a	3	3	3

(continued)

TABLE 8 (continued).

Parameter	Cd	Cr	Cu	Fe	Mn	Ni	Pb	Zn
Cool down °C	20	20	20	n/a	n/a	20	20	20
Ramp sec	1	1	1	n/a	n/a	1	1	1
Hold sec	19	19	19	n/a	n/a	19	19	19
Total sec/inj.	83	73	88	n/a	n/a	78	78	90

- * (1) sample in 5 ml 2N HNO₃ after 50:1 concentration with APDC/CoCl₂
 (2) sample in seawater with 10% HNO₃
 (3) sample in 1 ml 0.1N HNO₃ after evaporation and oxidation with H₂O₂ and second evaporation
 (4) sample in 2N HNO₃ with 10% seawater, 0.3M Ammonium phosphate matrix modifier
 (5) sample in 5 ml 2N HNO₃
 (6) sample in 5 ml 2N HNO₃, 10% seawater matrix modifier

** Total Zn time includes a 90 sec pause step for computer data reduction.

generally checked after every five samples had been atomized into the flame unit or injected into the HGA units. All samples were analyzed at least twice to determine signal reproducibility. Generally, for each 15 samples processed, one sample was determined by the method of standard addition, and one procedural blank sample was analyzed.

Fe, Mn, and Zn concentrations in extracts of seawater particulate samples were determined using the teflon cup FA-AA technique (Manning, 1975; Berndt and Slavin, 1978). The remaining elements (Cu, Cr, Cd, Ni, and Pb) were determined by HGA-AA. There was a slight matrix effect on Cd and Pb due to small quantities of salt on the filters which co-extracted with dissolved metals during elution with 5% HNO_3 . This matrix effect was overcome by modifying standards and samples with 10% seawater. Solutions were further modified by addition of ammonium phosphate. Matrix modification was performed in the graphite tube through use of a AS-40 autosampling device (Cooksey and Barnett, 1979) and a Stabilized Temperature Platform Furnace (STPF) (Slavin *et al.*, 1983).

Whole water analyses of Fe, Mn, and Zn were performed by direct injection into the HGA unit (Ediger *et al.*, 1974; Sturgeon *et al.*, 1979; Slavin, 1980). Mn determinations in dissolved seawater fractions were also determined by the direct injection technique (Fe and Zn were not determined in this fraction due to low signal-to-noise ratios). With the exception of adding an additional 100 μl of concentrated HNO_3 to 1-ml subsamples of the original acidified (pH 2-3) whole water and dissolved seawater samples, no chemical separation techniques were utilized to concentrate these elements from the seawater matrix. The large non-atomic background signal generated during atomization of sea salt was eliminated through use of the AA Model 5000 Zeeman background correction system (Fernandez *et al.*, 1980; Fernandez and Giddings, 1982). It was

necessary to matrix match unknown samples with standards since chemical interferences were not corrected by the Zeeman effect. Therefore, all standards were prepared in trace metal-stripped seawater and acidified in the same manner as the samples. Trace metal-free seawater was prepared using the methods of Davey et al. (1970).

Metals chemically separated and concentrated from seawater samples with the Co-APDC coprecipitation technique (Cu, Cd, Ni, and Pb) were determined with the model 603 AA and 2100 HGA units. It was not necessary to background correct these separated samples during atomization. However, samples and standards were matched. Standards were made in matrix solutions consisting of Co-APDC obtained from extraction of seawater blanks in a fashion identical to unknown samples. Spike recovery data for Cu, Cd, Pb, and Ni determined in blank seawater samples are given in Table 9.

All sediment and tissue elemental determinations (except Cd) were made using FA-AA. Cd concentrations in all predisposal and REFS sediment samples were too low to determine by FA-AA and had to be analyzed by HGA-AA. Spike recovery data for metals determined in sediment and mussel samples are given in Tables 10 and 11, respectively. Results obtained for mussel homogenate samples analyzed as a quality control check are given in Table 12. Use of this type of homogenate reference material has been described previously (Rogerson and Galloway, 1979; Galloway et al., 1983).

Organic Samples--

Figure 14 shows an overall flow diagram of the organic preparation and analysis scheme used in this study. There were differences in extraction procedures used for different sample types (e.g. whole water, foam plugs, sediment, or tissues) and these are described below. However, column chromatography

TABLE 9. MEAN RECOVERY OF TRACE METALS SPIKED INTO BLANK SEAWATER

Metal	Percent recovery	Standard deviation	N
Cu	106.3	9.2	37
Cd	98.4	6.5	37
Pb	99.9	8.6	17
Ni	102.9	5.2	5

TABLE 10. MEAN RECOVERY OF TRACE METALS SPIKED INTO SEDIMENT SAMPLE SOLUTIONS

Metal	Percent recovery	Standard deviation	N
Fe	103.2	9.5	3
Mn	92.3	1.0	2
Zn	105.1	---	1
Cu	103.1	5.8	4
Pb	103.9	6.7	8
Cd	108.9	8.9	5
Cr	115.0	24.9	8
Ni	98.2	7.2	4

TABLE 11. MEAN RECOVERY OF TRACE METALS SPIKED INTO MUSSEL SAMPLE SOLUTIONS

Metal	Percent recovery	Standard deviation	N
Fe	99.1	3.2	8
Mn	99.5	1.2	8
Zn	98.1	1.4	8
Cu	96.8	2.9	8
Pb	106.2	9.7	8
Cd	99.2	4.1	8
Cr	95.2	5.4	8
Ni	103.5	8.3	8

TABLE 12. TRACE METAL DETERMINATION OF MUSSEL HOMOGENATE SAMPLES

Metal	Mean ($\mu\text{g/g}$)	Standard error	Minimum	Maximum	N
Fe	429	23	406	452	5
Mn	24.0	0.6	21.7	28.3	5
Zn	126	7	120	128	5
Cu	12.8	0.9	11.5	13.6	5
Pb	9.3	1.3	7.8	11.4	5
Cd	2.6	0.2	2.0	2.6	5
Cr	2.9	0.3	2.2	2.9	5
Ni	7.8	0.6	7.0	8.4	5

ORGANIC ANALYSIS FLOW CHART

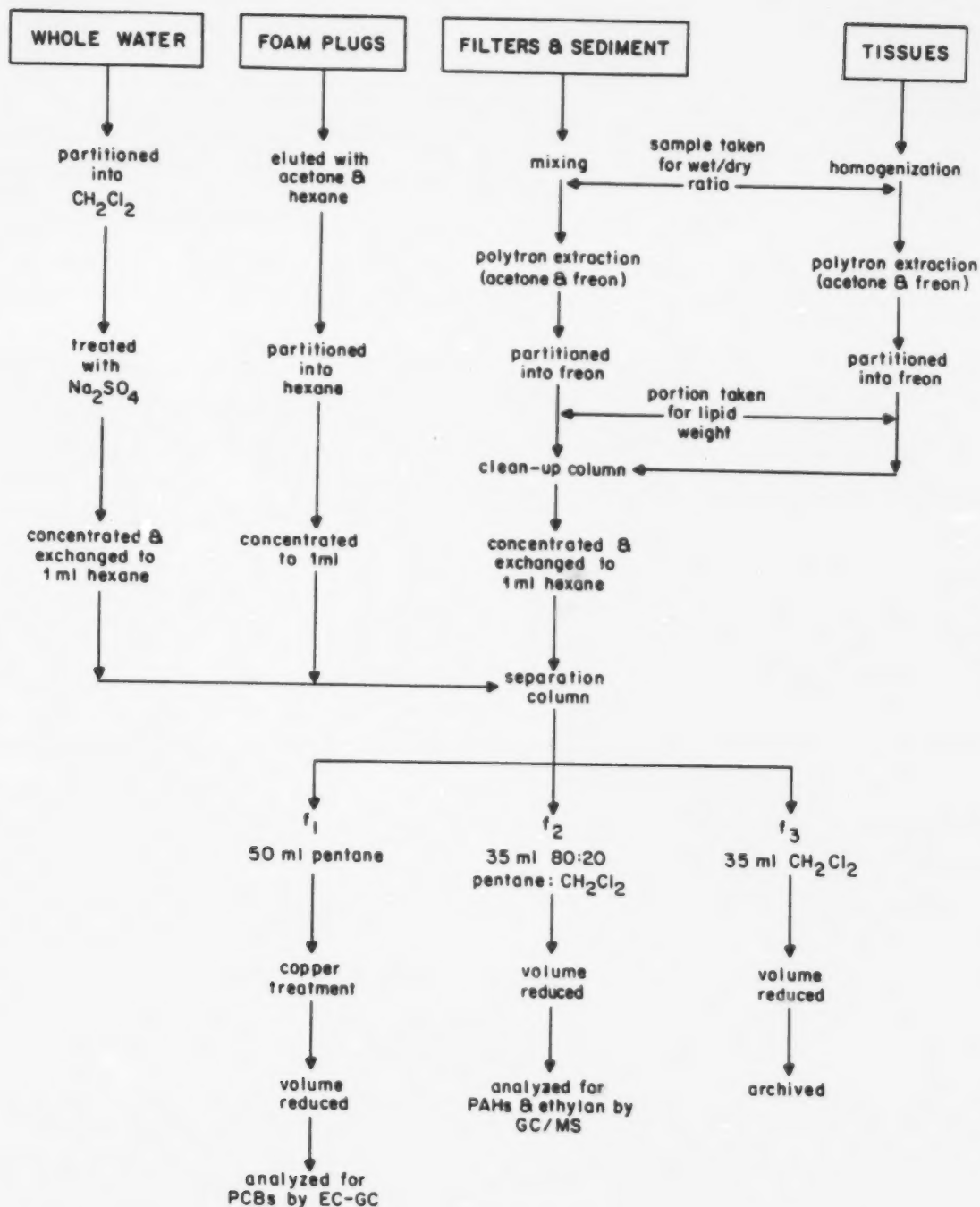


Figure 14. Organic chemical analysis flow chart.

chemical class separation methods were common to all samples. Data generated during preparation of samples (e.g. dry weight, wet weight, volume, etc.) were entered into a computerized data base for use in subsequent analyses (see Section 4 for a description of data bases and data management procedures).

All glassware used in sample analysis were washed with Alconox® and rinsed with tap water and deionized water, capped with aluminum foil and muffled at 450°C for 6 hours. They were stored capped with foil until used when individual items were uncapped and rinsed with an appropriate solvent.

Whole water extraction--Six 3.8-l containers (22.7 l total) of seawater were analyzed for each whole water sample. Water from each container and methylene chloride added at the time of collection were poured into a 6-l separatory funnel for extraction. An additional 100 ml of methylene chloride was added to bring the total volume up to 250 ml and the separatory funnel was shaken. After settling, the methylene chloride was drawn off the bottom, and 100 ml of methylene chloride was added to the funnel and the process was repeated. The methylene chloride extracts from all 6 bottles were combined in an Erlenmeyer flask and treated with sodium sulfate to remove traces of water. Sodium sulfate used in this step had been previously baked at 700°C for 8 hours, rinsed with methylene chloride and then placed in the Erlenmeyer flask. The sample was added and gently swirled, and poured into a round bottom flask leaving the sodium sulfate behind. The Erlenmeyer flask was rinsed, and the rinse combined with the sample.

Solvent volume was reduced to about 5 ml using a heating mantle below and a Kuderna-Danish evaporator and 3-ball Snyder column above the flask, and exchanged with an equivalent volume of hexane. This was transferred to a 10-ml concentrator tube fitted with a micro-Snyder column. An ebulator was added to each

tube, and the sample volume was reduced to approximately 0.8 ml in a tube heater with a foil cape. The ebulator was removed and rinsed with hexane, bringing the sample to 1 ml. At this point water sample extracts were placed on a silicic acid separation column. This procedure is described in the Column Chromatography section below.

Dissolved fraction extraction--Foam plug samples used to determine dissolved contaminant fractions were extracted by passing two 50-ml washes of acetone followed by two 50-ml washes of hexane through the teflon housing containing the foam plugs. These extracts were combined in a separatory funnel containing 150 ml of deionized water that had been pre-extracted with hexane. The solution was shaken, allowed to settle, and the hexane layer was drawn off. One hundred ml of hexane was added to this separatory funnel and the process repeated. The hexane extracts were combined and volume reduced as previously described for whole water samples, with the exception that foam plug extracts were not treated with sodium sulfate. These extracts were then ready for chemical class separations on the silicic acid column.

Particulate fraction and sediment extraction--Filter samples used to determine particulate contaminant fractions were placed into stainless steel centrifuge bottles for extraction. Sediment samples were mixed using a stainless steel spatula, and 1-3 g were removed to determine a wet to dry weight ratio. This portion was placed in a preweighed aluminum pan, weighed and dried to a constant weight at 95°C. Approximately 15-20 g of the original wet sediment were placed into a stainless steel centrifuge bottle for extraction.

Octachloronaphthalene was generally added at this point as a recovery standard. Twenty-five ml of acetone was added to each sediment or filter sample, and samples were homogenized at

25,000 rpm for 40 sec using a Brinkman Polytron with a stainless steel tip and brass bearings. After the first homogenization, samples were centrifuged at 10,000 rpm for 5 min in a refrigerated centrifuge at 4°C. The liquid phase (acetone-water) was decanted into a 1-l separatory funnel containing 150 ml of deionized water that had previously been extracted with freon (1,1,2-trichloro-trifluoroethane). A second 25-ml volume of acetone was added to the sample, and the homogenization and centrifugation repeated. The supernatant was again added to the separatory funnel. This procedure was repeated twice more using freon instead of acetone, and the freon extracts were also added to the separatory funnel. The funnel was shaken, and the freon layer was drawn off into a 500-ml Erlenmeyer flask. This process was repeated an additional two times using 50 ml of freon each time. The freon extracts were combined in the Erlenmeyer flask.

Total volume of freon was measured, and 20 ml was removed and placed into a preweighed aluminum pan. This solvent was allowed to evaporate, and the residue was dried to a constant weight at 90°C. The weight of residue was used to calculate total extractable lipid weight and the remaining extract was passed through a cleanup column. The column, which was 2 cm x 25 cm with a 500-ml reservoir, contained 50 g of Biosil A silicic acid (100-200 mesh, BioRad) that had been fully activated by heating at 120°C for 6 hours. The column was rinsed with 100 ml of methylene chloride followed by 100 ml of freon. The entire sample extract was added to the column and helium pressure was used to increase flow rate. After the sample level had reached the top of the column, 25 ml of freon and then 50 ml of methylene chloride were added to and eluted from the column. All of the column eluent was collected in a round bottom flask. This extract was then reduced in volume and the solvent exchanged with hexane and brought to 1 ml in a tube heater. The sample was then ready for fractionation on the separation column.

Tissue extraction--Tissue samples were homogenized using a Polytron with teflon bearings for 20 sec at 25,000 rpm. A portion of the homogenized sample was removed for determination of a wet to dry weight ratio (dried at 95°C to a constant weight). An appropriate amount of homogenate (usually 20 to 30 g wet weight) was added to a preweighed glass centrifuge tube, and sample weight recorded. Octachloronaphalene and 15 ml of acetone were added, and the sample was homogenized in the Polytron for 20 sec. The supernatant was drawn off into a separatory funnel containing 150 ml of deionized water that had been pre-extracted with freon. This polytron extraction was repeated once more with 15 ml of acetone and then twice using 25 ml of freon each time. The supernatants from each of these extractions were also added to the separatory funnel. The funnel was shaken and the freon was drawn off. This was repeated two more times using 50 ml of freon each time, and the freon extracts were combined. A measured portion of extract was removed for a total extractable lipid weight measurement, and the remaining portion was passed through a clean up column and reduced in volume as described for sediment and filter samples.

Column chromatography--Chemical class separations were achieved on a 0.9 x 45-cm column which contained 11.5 g of BioSil A silicic acid (100-200 mesh) that had been fully activated, and then 7.5% deactivated with water. This deactivation step was accomplished by adding an appropriate amount of water to the silica in a glass bottle and rolling it overnight. Consistent column activity was assured by testing each batch. A mix of PCB and PAH isomers were separated on a column made up of newly deactivated silica and the relative distributions of compounds in 3 column fractions were checked.

Before samples were added, the column was cleaned with 50 ml of methylene chloride and 50 ml of pentane. All sample extracts were added to a separation column in 1 ml of hexane

along with an additional container rinse of 1 ml of hexane. The first fraction (f1) was eluted from the column with 50 ml of pentane. The second fraction (f2) was eluted using 35 ml of 20% methylene chloride in pentane. A third fraction (f3) was then eluted with 35 ml of methylene chloride. The first fraction of sediment and filter samples was treated with activated copper powder to remove sulfur. Each fraction was then reduced in volume in a round bottom flask using a heating mantle and a Kuderna-Danish evaporator with a 3-ball Snyder column. The solvent was exchanged with approximately 5 ml of hexane and transferred to a 10-ml concentrator tube fitted with a micro-Snyder column. An ebulator and 0.8 ml of heptane were added to each tube, and the fractions were reduced in volume to 0.8 ml in a tube heater with a foil cape. The ebulator was removed and rinsed into the sample with heptane, bringing the volume to 1 ml.

The f1 fractions were analyzed by electron capture glass capillary gas chromatography (GC) for polychlorinated biphenyls (PCBs). The f2 fractions were analyzed for polynuclear aromatic hydrocarbons (PAHs) and associated compounds by gas chromatography-mass spectrometry (GC-MS). The f3 fractions contained relatively polar compounds such as phthalates and oxygenated PAHs. These fractions were archived and not analyzed as part of this study.

Gas chromatography--Isolated f1 fractions were analyzed for PCBs. For these analyses, 1 μ l of each sample was injected splitless into a Hewlett Packard 5840 gas chromatograph equipped with an electron capture detector and a 30 m DB-5 fused silica capillary column (J + W Scientific). Helium was used as the carrier gas at a flow rate of approximately 1.5 ml/min with the flow of a 95:5 mixture of argon:methane to the detector adjusted to 35 ml/min. Oven temperature was held at 80°C for 4 min and then increased from 80 to 290°C at 10°C/min.

Injector temperature was 270°C, while the detector was maintained at 300°C.

Concentrations were calculated as Aroclor® 1254 (A1254). The amount of A1254 was measured by comparing the sums of heights of 7 peaks in sample chromatograms to those of the same peaks in standards of A1254 that were analyzed at the beginning and end of each day. Calculations were performed by automated routines and stored in a computerized data base.

Quality assurance checks of gas chromatography analyses were made in several ways. Standards were analyzed at the beginning and end of each day. Results from these analyses were compared, and if the standard deviation of the two results was greater than 10%, all samples analyzed that day were reanalyzed. Standard curves were also periodically generated on each instrument to ensure that the response of the detector was linear over the range of concentrations measured. Recovery of standards was checked by spiking a known amount of octachloronaphthalene into each sample at the beginning of the sample analysis procedure. Recovery of this compound averaged 85±18% during this study.

Triplicate analyses of a sediment sample indicated that the precision of our PCB analysis procedure was 8.6% (coefficient of variation [CV]). A similar analysis of a triplicate mussel homogenate indicated a CV of 11%. The mussel homogenate used for this analysis had previously been analyzed as part of a Mussel Watch Program intercalibration study (Galloway *et al.*, 1983). This homogenate was also periodically analyzed during the course of this study and results of PCB and PAH analyses compared to earlier results in order to assure that analytical procedures were working properly.

Gas chromatography-mass spectrometry--The f2 fractions were

analyzed for selected polynuclear aromatic hydrocarbons (PAHs) and associated compounds using a Finnigan 4531 quadrupole GC-MS. Table 13 contains a listing of identifiable compounds along with molecular weights, dominant ions, and molecular formulas.

The Finnigan GC was operated with a capillary column in splitless injection mode. Each injection consisted of approximately 1 μ l of sample extract and approximately 2 μ l of solvent backflush. Split flow was approximately 50 ml/min and the septum sweep flow was approximately 2 ml/min. Both flows were suspended for 1 min just before sample injection. The DB-5 fused silica column was 30 m in length with a bore of 0.25 mm and a film thickness of 0.25 μ m. The GC oven was held at an initial temperature of 50°C for 2 min, ramped to 330°C at 10°C/min and held there for an additional 9 minutes.

The last 40 cm of the column passed through the transfer line oven area, which was heated to 300°C, and then to within 1 cm of the source volume in the source area of the MS, which was maintained at 150°C. The source was operated in electron impact mode at 70 ev. Filament emission current was maintained at 200 μ amps and the various source potentials were adjusted to produce a spectrum of decafluorotriphenylphosphine (DFTPP) which would meet specifications detailed by Eichelberger *et al.* (1975). The MS was scanned from 15 to 650 amu in 1 sec while collecting 10 centroid samples per peak. The continuous dynode electron multiplier was operated near 1,550 volts, and preamplifier sensitivity was set to 10^{-8} amps/volt. Raw data for each sample were stored on disk.

On each day that samples were to be analyzed, the mass scale of the MS was calibrated by emitting perfluorotributylamine into the source, acquiring data, and

TABLE 13. POLYNUCLEAR AROMATIC HYDROCARBONS AND RELATED COMPOUNDS QUANTIFIED USING GC-MS

Molecular weight	Molecular formula	Major ion	Compound	Text abbreviation
166	C ₁₃ H ₁₀	166	Fluorene	P0S166
180	C ₁₄ H ₁₂	180	C1 Homologs of Fluorene	H1C166
194	C ₁₅ H ₁₄	194	C2 Homologs of Fluorene	H2C166
208	C ₁₆ H ₁₆	208	C3 Homologs of Fluorene	H3C166
222	C ₁₇ H ₁₈	222	C4 Homologs of Fluorene	H4C166
184	C ₁₂ H ₈ S	184	Dibenzothiophene	P0S184
198	C ₁₃ H ₁₀ S	198	C1 Homologs of Dibenzothiophene	H1C184
212	C ₁₄ H ₁₂ S	212	C2 Homologs of Dibenzothiophene	H2C184
226	C ₁₅ H ₁₄ S	226	C3 Homologs of Dibenzothiophene	H3C184
240	C ₁₆ H ₁₆ S	240	C4 Homologs of Dibenzothiophene	H4C184
178	C ₁₄ H ₁₀	178	Phenanthrene	P1A178
178	C ₁₄ H ₁₀	178	Anthracene	P2B178
192	C ₁₅ H ₁₂	192	C1 Homologs of Phenanthrene and Anthracene	H1C178
206	C ₁₆ H ₁₄	206	C2 Homologs of Phenanthrene and Anthracene	H2C178
220	C ₁₇ H ₁₆	220	C3 Homologs of Phenanthrene and Anthracene	H3C178
234	C ₁₈ H ₁₈	234	C4 Homologs of Phenanthrene and Anthracene	H4C178
202	C ₁₆ H ₁₀	202	Fluoranthene	P1A202
202	C ₁₆ H ₁₀	202	Pyrene	P2B202

(continued)

TABLE 13 (continued).

Molecular weight	Molecular formula	Major ion	Compound	Text abbreviation
216	C ₁₈ H ₁₂	216	C1 Homologs of Fluoranthene and Pyrene	H1C202
230	C ₂₀ H ₁₄	230	C2 Homologs of Fluoranthene and Pyrene	H2C202
244	C ₂₁ H ₁₆	244	C3 Homologs of Fluoranthene and Pyrene	H3C202
258	C ₂₂ H ₁₈	258	C4 Homologs of Fluoranthene and Pyrene	H4C202
228	C ₁₈ H ₁₂	228	Benz(a)anthracene	P1A228
228	C ₁₈ H ₁₂	228	Chrysene	P2B228
228	C ₁₈ H ₁₂	228	Sum of all MW 228 parent PAHs	P0S228
242	C ₁₉ H ₁₄	242	C1 Homologs of all MW 228 parent PAHs	H1C228
256	C ₂₀ H ₁₆	256	C2 Homologs of all MW 228 parent PAHs	H2C228
270	C ₂₁ H ₁₈	270	C3 Homologs of all MW 228 parent PAHs	H3C228
284	C ₂₂ H ₂₀	284	C4 Homologs of all MW 228 parent PAHs	H4C228
252	C ₂₀ H ₁₂	252	Sum of Benzofluoranthenes	P1A252
252	C ₂₀ H ₁₂	252	Benzo(e)pyrene	P2B252
252	C ₂₀ H ₁₂	252	Benzo(a)pyrene	P3B252
252	C ₂₀ H ₁₂	252	Perylene	P4B252
266	C ₂₁ H ₁₄	266	C1 Homologs of all MW 252 parent PAHs	H1C252
280	C ₂₂ H ₁₆	280	C2 Homologs of all MW 252 parent PAHs	H2C252
294	C ₂₃ H ₁₈	294	C3 Homologs of all MW 252 parent PAHs	H3C252
308	C ₂₄ H ₂₀	308	C4 Homologs of all MW 252 parent PAHs	H4C252
276	C ₂₂ H ₁₂	276	Sum of Benzo(ghi)perylene, Anthanthrene and others	P0S276

(continued)

TABLE 13 (continued).

Molecular weight	Molecular formula	Major ion	Compound	Text abbreviation
278	C ₂₂ H ₁₄	278	Sum of Picene, Dibenzanthracenes and others	P0S278
300	C ₂₄ H ₁₂	300	Coronene	P0S300
302	C ₂₄ H ₁₄	302	Sum of Dibenzochrysenes, Dibenzonaphthacene and others	P0S302
306	C ₁₈ H ₂₀ Cl ₂	223	Ethylan	P0S223

running a software calibration routine. Following calibration, a solution of standards was analyzed to determine the response of standards, their retention time, and the spectrum of the DFTPP. If any of these determinations were outside of predetermined limits, remedial action was taken. Prior to each sample run, the sample extract was reduced in volume to a level which was thought to provide adequate sensitivity. After completion of each run, major peaks were checked to ensure that the sensitivity was adequate and that no saturation had occurred. After all samples for the day had been run, another standards run was made.

Quantification with GC-MS involved using the data system to measure areas of selected peaks using extracted ion current profiles (EICPs). An EICP is a trace of the intensity of the selected fragment ion with respect to time. The selected fragment is usually the molecular ion of the compound of interest. This area measurement is not truly an area measurement, but rather a sum of the heights above baseline of the EICP at the 4 or 5 points measured during elution of the compound of interest. In all cases, peaks were manually integrated, that is, baselines were visually determined and heights measured by hand from the EICPs displayed on the terminal screen. This better enabled the operator to check that the compound had eluted at the correct time and without interference. For a number of EICPs, known interferences were also measured so that they could be subtracted later. A quantification list was compiled for each sample or standard run which contained areas of the various peaks of interest as determined from their respective EICPs.

Quantification was accomplished by the method of internal injection standards (IISs). Two IISs, monofluorobiphenyl and monofluorofluoranthene, were added to sample extracts just prior to injection. The standards solution also contained these two

IISs. Standards runs were used to determine Relative Response Factors (RRFs); i.e., RF of the standard divided by RF of the IIS, where RF is the Response Factor and is defined as the area counts of the peak divided by nanograms injected. Actual concentrations of specific compounds in the sample were then calculated by using the RF of the IIS in the sample itself, the area of the compound in the sample, the known amount of IIS added to the sample, and the dry weight (or volume, if a water sample) of the original sample. A plot of concentrations of selected compounds versus molecular weight was examined. The distribution of compounds was carefully checked to ensure consistency with previous data. Only after this quality assurance step was performed were data added to the permanent computerized data base. Raw GC-MS data were archived on magnetic tape.

Total carbon analysis--Two to five g of wet sediment were placed in an aluminum pan and dried at 90°C. After drying, the hardened sediment was carefully broken apart with a dissecting needle and all shell debris (>1 mm) was removed with metal forceps. The sample was ground to a fine powder using a mortar and pestle and transferred to a 20-ml scintillation vial for storage until analyzed.

These samples were analyzed using a Carlo Erba Model 1106 CHN Analyzer. A small amount (5-10 mg) of sample was weighed into a tin boat using a Cahn Model 4700 electrobalance. Appropriate amounts of a standard (acetanilide) were also weighed into tin boats. Empty tin boats served as blanks for determining background contamination. Samples and tin boat blanks were crimped into small balls and introduced into a combustion tube that was maintained at 1,040°C. The sample was combusted at 1,500°C. Data were collected from this instrument using a Hewlett Packard Model 3390A integrator.

Concentrations were calculated using a standard curve generated from analysis of acetanilide standards and corrected for blank values. As a quality assurance procedure, separate acetanilide samples were routinely analyzed. Results were compared to values predicted by the standard curve. Total carbon concentrations were manually entered into a computerized data base for future analysis.

Results of these analyses are referred to as total carbon although the bulk of carbon present was organic. Some sediments contain significant amounts of calcium carbonate (CaCO_3), which may necessitate treatment with acid in order to remove this carbon as CO_2 . However, this is generally not a significant problem with nearshore sediments if obvious shell material is removed from samples. In fact, Aller (1982) indicated that even if shell material was not removed, CaCO_3 content could only account for less than 20% of carbon present in a Central Long Island Sound sediment core. His sediment core was collected from a location very close to our study area.

In order to quantify the contribution of CaCO_3 to the total carbon measurement, test sediment samples were acid treated prior to analysis. This involved adding 1N HCl to 1-2 g of sediment in a glass scintillation vial and allowing it to sit for 6 hours. The sample was then heated at 55°C until dry (approximately 3 days). Some problems were encountered with this procedure, particularly for sediments with very high organic carbon concentrations (e.g. Black Rock Harbor sediment). The acid desorbed large amounts of organic material from particles and during drying this material coated the sides of the sample container. It was not possible to remove completely this material from container walls and mix back into the sample. However, samples analyzed for total carbon before and after acid treatment indicated that measured total carbon concentration decreased at most by 21% due to acid treatment (see Table 14).

TABLE 14. TOTAL CARBON CONCENTRATION (MG/G DRY WEIGHT) IN SEDIMENTS BEFORE AND AFTER ACID TREATMENT

Sediment	Before treatment*	After treatment*	% Difference
REFS (0-2cm)	23.40	19.77	15
200E (0-2cm)	24.65	20.44	17
CTR (0-2cm)	26.84	21.14	21

* mean of 2 replicate analyses

TABLE 15. CONTRIBUTION OF CALCIUM CARBONATE CARBON TO TOTAL CARBON IN SEDIMENTS AS MEASURED BY MICROBOMB CALORIMETRY

Sediment	CaCO ₃ *	CaCO ₃ carbon*	Total carbon*	% CaCO ₃ carbon
REFS	4.2	0.504	25.76	2.0
CTR	1.6	0.192	45.68	0.4
BRH	4.3	0.516	76.00	0.7

* mean of 2 replicate analyses, in mg/g dry weight

Test samples were also analyzed for CaCO_3 by micro-bomb calorimetry by J. Boucher at the University of Rhode Island's Graduate School of Oceanography. The technique used was similar to Aller (1982). A 1-g sample was acidified with 10% HCl inside a sealed bomb, and the total volume of gas evolved was measured as a pressure increase using a capillary manometer. These results are shown in Table 15 and indicate that only 0.4-2% of carbon present in the samples was due to CaCO_3 .

The reason for the discrepancy between acid treatment and microbomb estimates of the contribution of organic carbon to total carbon is not known; however, it is possible that acid treatment may oxidize some of the organic matter or that some of the liberated organic matter was not quantitatively transferred from the sides of the container back into the sample after acid treatment. Micro-bomb calorimetry was not available for all samples and because of problems encountered with acid treatment this technique was not used. Therefore, results are reported as total carbon after removal of visible shell material.

SECTION 4

DATA MANAGEMENT

GENERAL HANDLING AND STORAGE OF DATA

Data used in support of the exposure assessment component of the FVP were gathered from a variety of sources. Because of this, several media were used to store the data. For instance, field chemistry data and physical parameter information generated by ERLN personnel were entered into computerized data bases by hand or through automated procedures. Other information, such as National Weather Service data and the physical parameter data collected by in situ instrumentation were transferred to ERLN on 9-track magnetic tape. These data were read into active disk files for manipulation and analysis. Journal articles, government reports, and reports solicited by EPA provided data which were transcribed to disk files by hand. The data gathered and encoded by all of these methods were verified for accuracy prior to their use in exposure assessment analyses.

Much of the exposure assessment data were stored as disk files on ERLN's PDP 11/70 computer system. The diverse nature of these data precludes more detailed descriptions of the formats of individual data files. With the exception of information generated during ERLN field cruises, most of the data resided in FORTRAN accessible data files. Hard copy records of these files were maintained for back-up purposes. A description of the data base system developed in part to store data generated by ERLN's exposure assessment field work is given below.

MANAGEMENT OF ERLN'S EXPOSURE ASSESSMENT FIELD DATA

DATMAN Data Management System

Field data collected by EPA are organized and stored on ERLN's PDP 11/70 computer system as a series of data bases (DBs). Data entry, correction, retrieval, and manipulation were achieved through use of the data management utility DATMAN (Bass, 1980, 1982), a system of management and elementary analysis routines. Control of DATMAN is accomplished through interactive sessions by the user, or through the use of batch processing.

A full DB in DATMAN is maintained as six individual files used to store and organize the data. Five of these files contain information concerning the names of the files and the meaning and formats of variables used in the DB. The sixth file contains the actual data, which may be integer (stored internally as 4 byte integer), real (stored as 8 byte real), or alphanumeric (stored in ASCII) in nature. DATMAN provides routines for creation and maintenance of each of these files. Active DBs are stored on the PDP 11/70 as disk files for routine use. Weekly back-up procedures are performed by ERLN's Automated Data Processing (ADP) support staff to minimize the loss of data due to system malfunction or user error.

Data residing in DATMAN DBs are organized logically into tabular form in which rows or records correspond to single observations and columns correspond to attributes or variables. Each record contains values for all variables within the DB. Certain of these variables contain information which uniquely identifies individual observations, such as the location of sample collection in time and space. These variables are called primary index variables. One primary index variable associated with all FVP samples is the variable SAMPNUM, a six digit sample

number uniquely assigned to each sampling unit as it was entered into the DB system. In addition to primary index variables, other variables may be used to further delineate a sample, and to aid in retrieval of subsets of records. The measured quantities of interest, such as suspended solids concentration, may reside in one or more variables in the DB. Generally, comment variables are included for any special information not described by primary and secondary index variables. All DBs also contain variables used for record keeping purposes, such as those describing the time and date of record entry.

Two DBs containing a variable in common may be logically joined together by association of data in one DB with records in the second DB. This is accomplished by matching those records with equivalent values of the common variable residing in the two DBs. A logical DB resulting from such a join may then be joined with a third DB, and so on. Generally, the variable used to accomplish a join is a primary index variable, such as SAMPNUM, although records may also be associated by the values of non-index variables specified by the user. As information gathered from a single sample may reside in several different DBs, primary index variables may be used to bring together all information regarding that sample.

The wide variety of information obtained during this project necessitated the creation of several DBs which organize the data in a logical fashion. Thus, chemistry data reside in a series of DBs containing organic and trace metal contaminant concentrations, sample preparation information, and so on. Physical parameter data reside in a different series of DBs. A general overview of routes of information transfer and the relationships between DBs associated with the exposure assessment component are presented in Figure 15. Brief descriptions of information found in each of these DBs is given in the following sections.

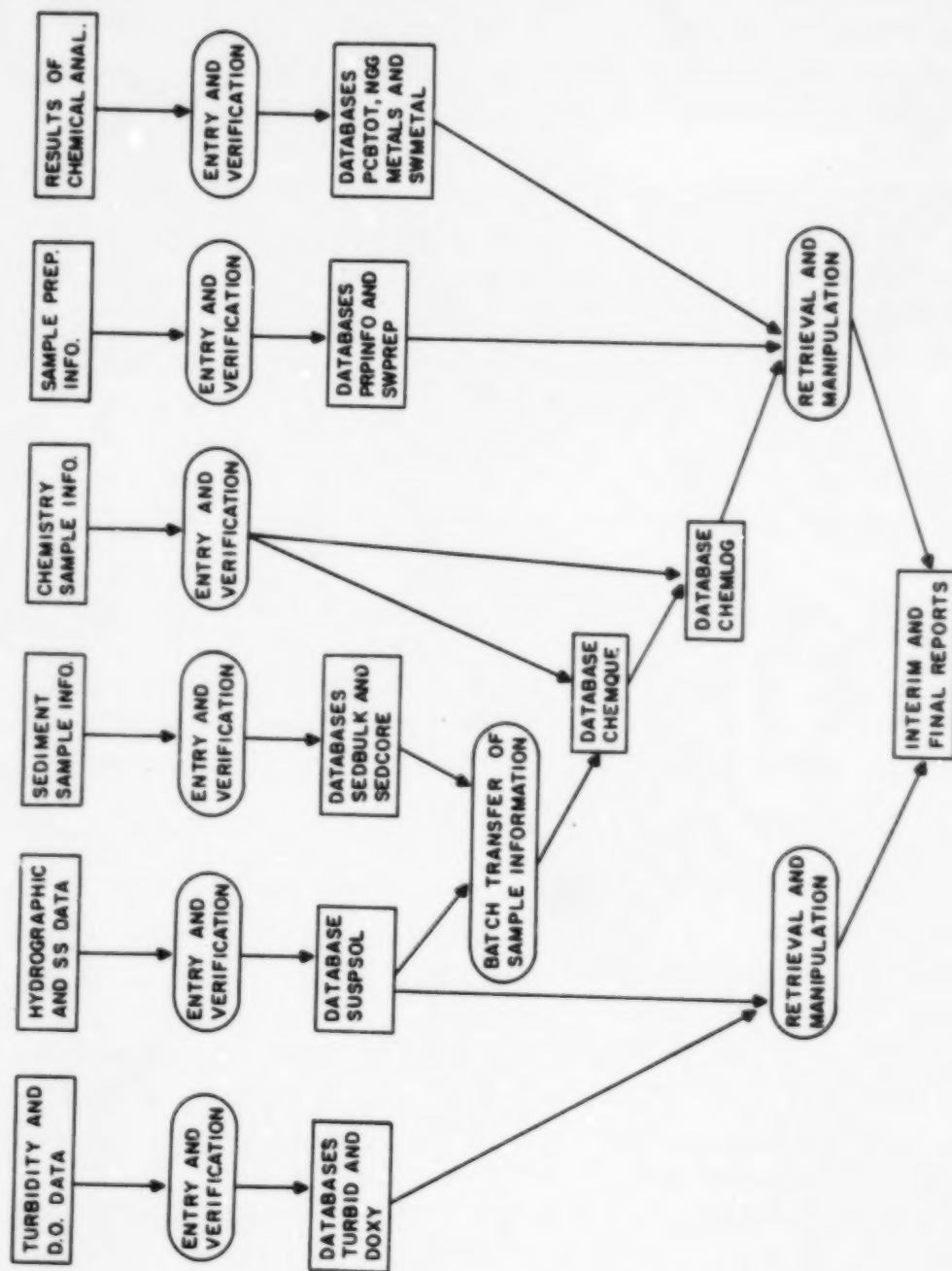


Figure 15. Logical relationship and flow of data between the physical parameters and chemistry data base systems.

Chemistry System

The chemistry DB system was designed by the ADP support staff (Rosen and Sherman, 1984) to facilitate handling and storage of the large amounts of data produced by the FVP. Unlike the physical parameter DBs described later, individual chemical concentration DBs contain the variable ID along with combinations of the variables REP, STUDY, and RUN (or RN) as sole primary index variables. ID is a unique sample number assigned to a sample upon receipt by the chemistry staff. STUDY is an alphanumeric code designating the project from which a sample originated (for instance, STUDY = 'FV' for the FVP). The variable REP is intended to identify replication in the analysis of a sample (except in the DB SWMETAL, where it is redundant in meaning with ID). No information concerning collection or identification of a sample follows the sample or is stored in these DBs. The dissociation of descriptive sample information from the reported contaminant concentrations permitted chemical analyses to be carried out in a blind fashion.

Sample identification information is stored in the DB CHEMLOG, which may then be joined via the variable ID with the concentration DBs to obtain a full description of a sample. Sample information is entered into CHEMLOG in either of two ways: via direct entry by members of the chemistry group, or via a series of batch routines initiated by outside users which first transfer data to a repository DB named CHEMQUE. In addition to the variable ID, CHEMLOG also contains information concerning the time and location of sample collection, the origin of the sample, the types of chemical analyses to be performed, and the primary index variable SAMPNUM. A full listing of variables found in CHEMLOG is presented in Appendix B, Table B-1.

Working copies of chemistry DBs are located in directories

unavailable to the general user community. Straight copies of these DBs are made on a weekly basis to a directory designated for general use. This procedure allows some quality control on the data being released for statistical analysis and use by other researchers.

Chemistry Data Bases

Five DBs are maintained to manage contaminant concentration data obtained from sediment, water column, and biotic field samples. Information concerning organic contaminant concentrations are stored in the DBs PCBTOT and NGG (Appendix B, Tables B-2 and B-3). PCBTOT contains the measured concentrations of total PCBs quantified as Aroclor® 1254. NGG contains the concentrations of 43 PAH parent and alkyl homolog compounds (listed in Appendix B, Table B-3), as well as partial sums of those parent and homolog compounds with common molecular weights.

Trace metal concentrations measured in water column samples are stored in SWMETAL. Similarly, trace metal concentrations found in sediment and biotic samples are stored in the DB METALS. A list of the metals for which chemical analyses are performed, as well as other variables in these two DBs may be found in Appendix B, Tables B-4 and B-5.

The fifth DB, CARBTOT, contains values for total carbon measured in sediment samples. The primary index variables in CARBTOT are ID and REPCARB. Thus, the association of sample collection information in CHEMLOG with total carbon measurements is accomplished by joining the two DBs using ID. A full description of the variables found in CARBTOT is presented in Appendix B, Table B-6.

In addition to CHEMLOG and the concentration DBs described

above, the DBs PRPINFO and SWPREP are maintained for information concerning preparation and analysis of chemistry samples. These two DBs, as with all of the chemistry DBs, may be joined with CHEMLOG or the concentration DBs via the variable ID. Additional variables found in PRPINFO and SWPREP are described in Appendix B, Tables B-7 and B-8.

Physical Parameter Data Bases

Physical parameter information collected during cruises to the FVP study area are organized and stored in three DBs. Brief descriptions of variables found in these DBs are given in Appendix B, Tables B-9 through B-11. In addition to the variable SAMPNUM, all three of these DBs contain the primary index variables DATE, STATION, HOUR, DEPTH, and REP. Thus, each record contains a complete description of the sample, so that joining to a second DB for this purpose becomes unnecessary.

Suspended solids concentrations measured in the water column are stored in the DB SUSPSOL. Temperature, conductivity, and salinity profiles are also stored in this DB. Sampling information for measurements corresponding to trace metal samples are ultimately dropped to CHEMLOG from SUSPSOL via a series of batch routines for association with the concentration data in SWMETAL. Turbidity profiles are stored in TURBID, which also contains those suspended solids measurements which were obtained concurrently with the light transmission data. Dissolved oxygen profiles collected during physical parameter cruises and measurements made at single depths at other times are stored in the DB DOXY.

Two similarly constructed DBs contain information concerning sediment samples (Appendix B, Tables B-12 and B-13). These DBs are used primarily for sample tracking and identification. SEDBULK describes sediment scoop samples

collected prior to and immediately subsequent to the disposal operation. SEDCORE contains similar information for samples collected as cores from Smith-MacIntyre grabs. Identification information is ultimately dropped from both of these to CHEMLOG for association with sediment chemistry information found in METALS, PCBTOT, and NGG.

SECTION 5

QUALITY ASSURANCE/QUALITY CONTROL

The objective of the ERLN Quality Assurance (QA) program is to ensure and document that all data collected, stored, reported, and/or used by ERLN are scientifically valid, defensible, and are of known precision, accuracy, completeness, comparability, and representativeness. The rationale and procedures for meeting this objective are delineated in the ERLN QUALITY ASSURANCE PROGRAM document, Fifth Revision -- Document ERLN/QA-5, available from the ERLN QA Officer. All research activities at ERLN must be performed in conformity with an approved research protocol which is filed with the Division Director prior to commencement of the project. This protocol must include the QA Project Plan, which includes research hypotheses and how they are to be tested.

All data generated as a part of an approved research project must meet a number of requirements, which include using approved analytical procedures and/or SOPs where applicable; the reporting of precision, accuracy, representativeness, and completeness of all data; development of the intended use(s) and acceptance criteria of data prior to its collection; and the collection of environmental data at a level of quality necessary for the intended use. Data processing procedures must be documented, reviewed, and revised as required to meet data quality requirements. Data will be validated according to specified and approved criteria.

The Project Officer or Principal Investigator is

responsible for developing and implementing the research protocol including the QA Project Plan. Laboratory management is responsible for including evaluation of these quality assurance programs during execution and reporting. At ERLN the QA Project Plan takes the form of a QA Questionnaire and its necessary appended materials. The Project Questionnaire is divided into Task-level components encompassing:

1. statistical basis of experimental design and sampling plan;
2. quality control of organisms studied;
3. quality control of field implementation;
4. quality control of laboratory implementation;
5. quality control of physical and chemical parameters; and
6. quality control of recordkeeping.

A completed Project Questionnaire for the FVP Exposure Assessment Program was prepared by and is available from Dr. John F. Paul, Principal Investigator. Where appropriate, specific quality assurance procedures and activities have been described within the body of this report.

SECTION 6

DATA PRESENTATION AND INTERPRETATION

PHYSICAL PARAMETERS

The following presentation of information gathered during this study concerning physical characteristics of the sediments and water column, atmospheric weather conditions, and freshwater input to the Sound focuses on observed values and the sources of spatial and temporal variability. Where appropriate, general descriptions of Sound-wide patterns are given to help put data specific to the FVP study area into perspective. Examples of data collected from in situ instrumentation array deployments and from REMOTS® benthic camera surveys are offered where they aid in the understanding of physical processes. Complete descriptions of these data may be found in Bohlen and Winnick (1984a,b, 1986), Bedford et al. (1986a,b) and Scott et al. (1984, 1987).

Weather Data

General weather patterns affecting LIS are typical of those found in temperate regions of the Northern Hemisphere. Atmospheric fronts approach chiefly from the west and southwest. The frequency and intensity of major storm events increase during winter months, whereas summers are generally more calm. These weather patterns in part drive the physical parameters measured in the water column, as well as influence the resuspension and transport of bottom sediments.

Daytime air temperature measured in 1983 at the Bridgeport, CT, National Weather Service Station can be seen in Figure 16. Peak temperatures occurred during the summer months of June through September, with highs approaching 35°C. Minimum temperatures near -18°C occurred in December, January, and February. Diel variation of up to 7°C was observed at various times of the year. Larger scale variation (on the order of weeks), associated with the passage of atmospheric fronts, is also apparent in this figure. Daily temperature records are reported in Appendix C.

Figures 17 and 18 show sea level atmospheric pressure and wind speed measured at Bridgeport in 1983. Storm events are indicated by periods of low atmospheric pressure and high wind speed. The summer months following the FVP disposal operation were relatively calm, with minor storms occurring in late June and mid-July. Wind speeds during these events were less than 14 m/sec. Larger storms did not occur until mid-November, when wind speeds exceeded 15 m/sec. The lowest atmospheric pressure recorded in 1983 occurred in early December (Figure 17), and was accompanied by wind speeds reaching 18 m/sec (approximately 40 mi/hr). Since maximum fetch direction in LIS is oriented primarily east-west, winds blowing along this axis have maximum influence on wave activity and subsequent effects on bottom sediment resuspension. Wind direction during the December storm event was primarily from the south and southwest (Figure 19). Thus the potential impact of this storm on sediment resuspension was lessened somewhat. A more complete analysis of the influence of post-disposal atmospheric weather conditions on wave activity and bottom resuspension will be given in a subsequent report. Climatological data are given in Appendix C.

Surface Runoff

Freshwater input into LIS occurs primarily from

BRIDGEPORT 1983

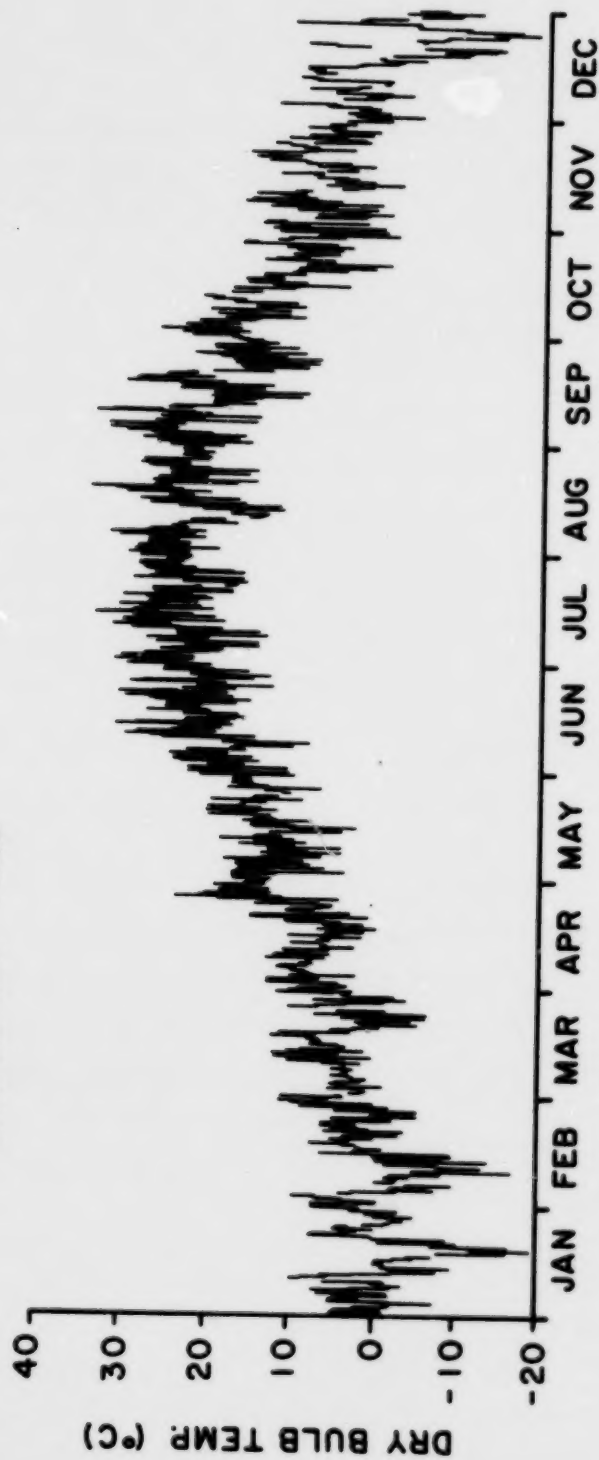


Figure 16. Daytime air temperature at Bridgeport in 1983.

BRIDGEPORT 1983

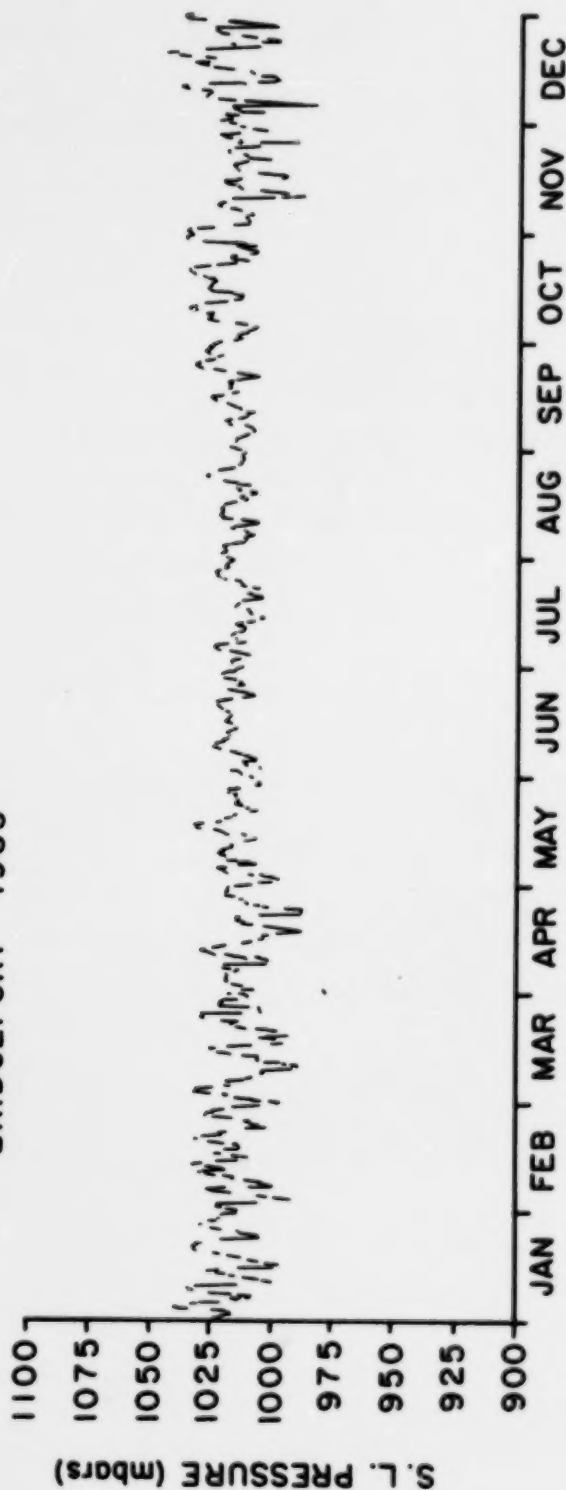


Figure 17. Sea level atmospheric pressure at Bridgeport in 1983.

BRIDGEPORT 1983

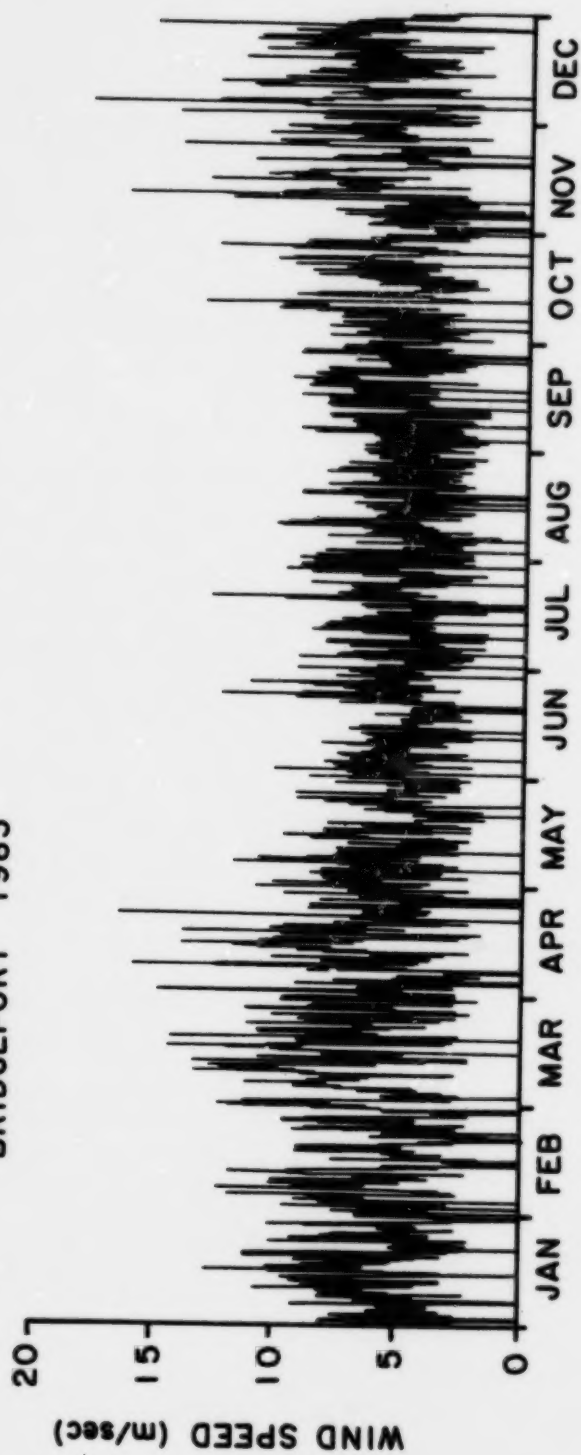


Figure 18. Wind speed measured at Bridgeport in 1983.

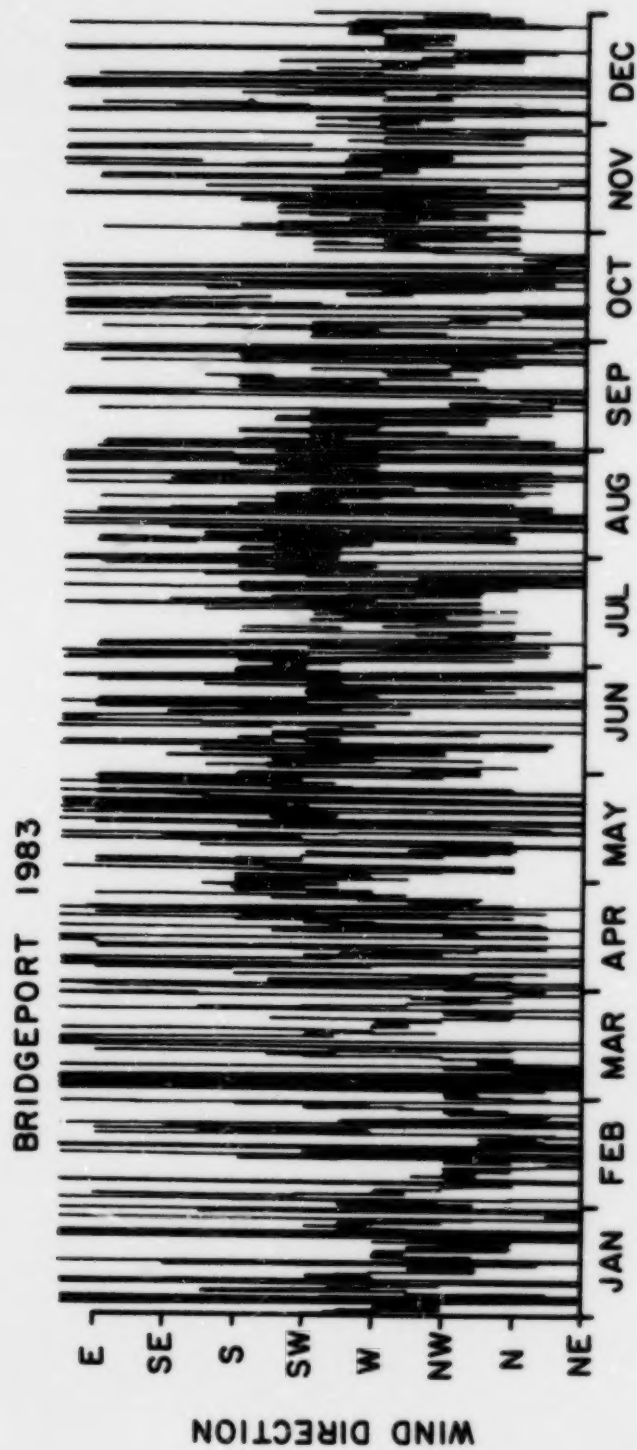


Figure 19. Wind direction measured at Bridgeport in 1983.

precipitation directly into the Sound and runoff from major drainage basins located on surrounding land masses. Seasonal patterns of precipitation can be deduced from surface runoff data.

As few rivers drain into the Sound from Long Island, NY, the major sources of riverine input affecting CLIS derive from those along the Connecticut coast. Monthly stream discharge of the largest of these, the Housatonic, Quinnipiac, Connecticut and Thames Rivers, is summarized in Figure 20. Annual freshwater input from these systems was approximately $1.2 \times 10^{13} \text{ ft}^3$ during the 1983 calendar year, with somewhat lesser amounts entering the Sound in 1982 and 1984. Runoff data for 1985 were not available as of this writing. Periods of highest flow occurred during March through June as a result of snow melt and spring rains. Approximately 65% of annual runoff occurred during this spring freshet. An interesting anomaly occurred in December 1983, when runoff increased dramatically relative to the previous month. This mean flow rate reflects the increased storm activity recorded during that month. Stream flow was relatively low from July through October in all three years, reflecting seasonal decreases in precipitation. Smaller river systems around the Sound were assumed to display similar seasonal patterns of discharge.

Current and Tidal Data

The tidal regime of LIS is chiefly diurnal in nature. Bottom topography and the orientation of land masses around the Sound result in tidally-induced current flow oriented primarily along the east-west axis. Although some water exchange takes place near Throg's Neck at the western end of the Sound, the majority of exchange occurs between Plum and Fisher's Islands (The Race) at the eastern end. Current speeds in The Race are high, as cooler, more saline shelf waters enter on flood tides

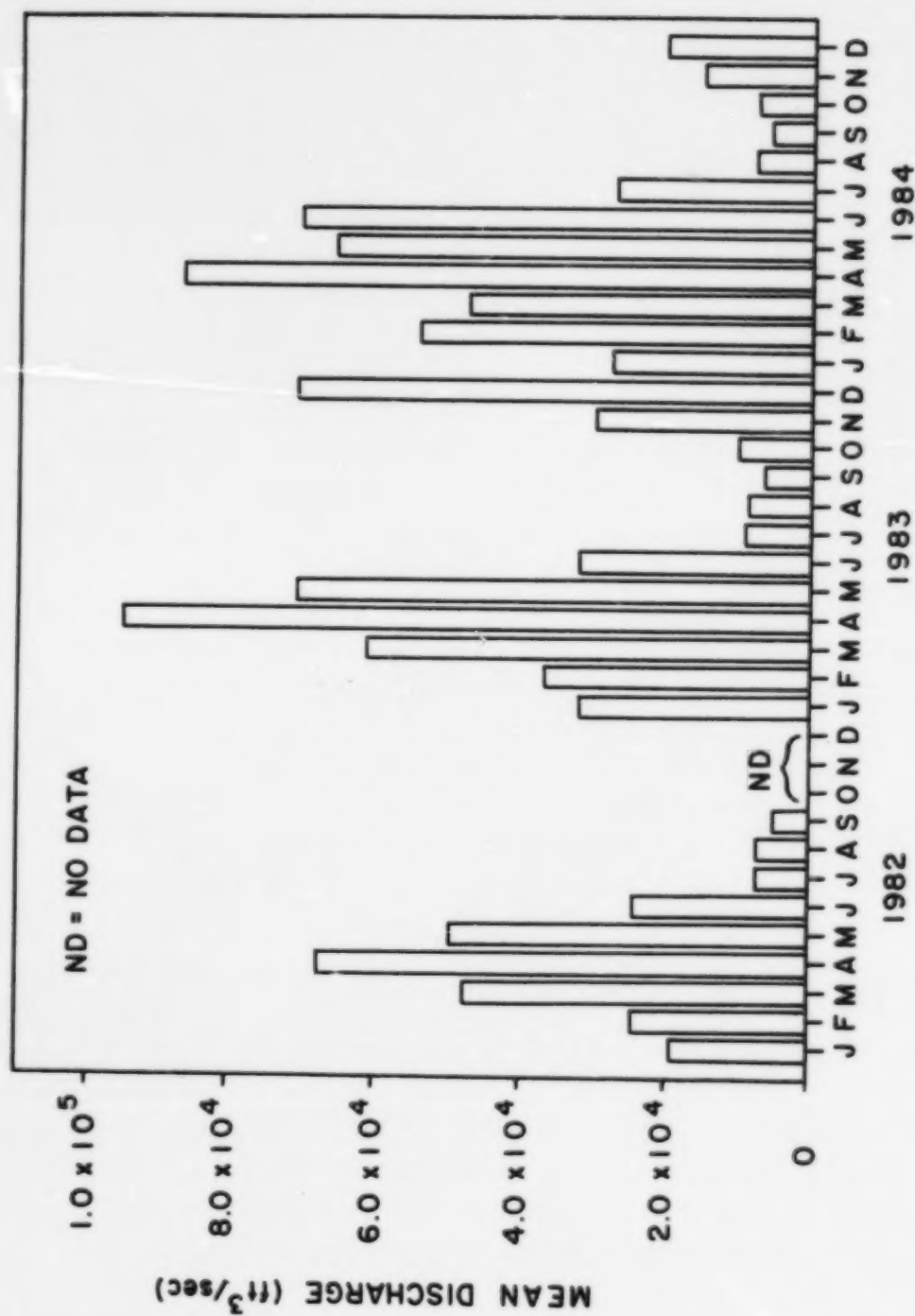


Figure 20. Monthly stream discharge data from major rivers in Connecticut.

and are mixed with Sound waters. These waters leave the Sound during periods of ebb flow. Maximum tidal speeds in the Sound approach 5 knots near The Race, while in the western end of the Sound they reach a maximum of 1.4 knots (LeLacheur and Sammons, 1932; Riley, 1956). In addition to the general east-west transport of water, float releases indicate long-term, near-bottom drift in a northwesterly direction (Gross and Bumpus, 1972). These generalized patterns of current flow within LIS have implications in the discussion of Sound-wide material transport and the structure of bottom sediment distributions (see below). A discussion of the general estuarine circulation found in the Sound and the factors which influence this is given by Riley (1956).

The FVP study area is characterized by tidal fluctuations averaging approximately 2 m in height (Long, 1978). Current speeds near the surface reach 0.75-1.0 knots during periods of maximum ebb and flood tidal flow. Figure 21 shows speeds measured in the east-west direction at approximately 1 m above bottom at 1000E during the 17 April 1984 deployment of the DAISY in situ instrument array. Maximum speeds approached 0.6 knots during peak tidal flow. This value is somewhat higher than the 0.3 knots measured along the east-west direction at 1 m above bottom by Rhoads and Boyer (1982) at a nearby station. The difference may be due to the depth of the water column, as their station resided at a depth of 15.5 m compared with the 20 m of 1000E. Current speeds measured in the north-south direction during the 17 April 1984 deployment are shown in Figure 22. Speeds along this axis approached 0.4 knots. Based upon these measurements, a parcel of water could be transported several kilometers downstream over the course of a single ebb or flood tide.

Small scale variation superimposed on the tidal patterns of currents in Figures 21 and 22 resulted from internal waves in

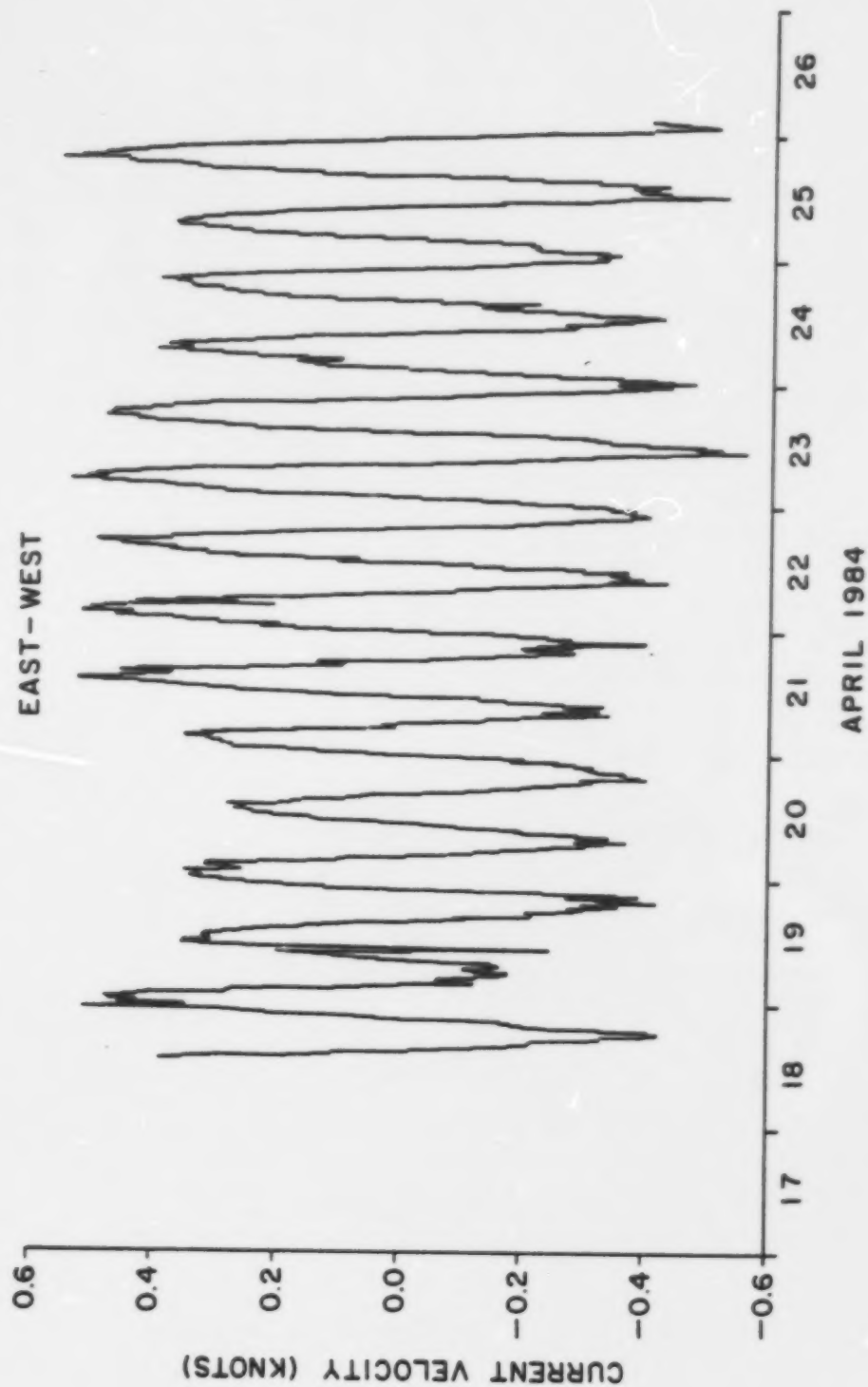


Figure 21. East-west current velocity at 1 m above bottom during the 17 April 1984 deployment of the in situ instrumentation array.

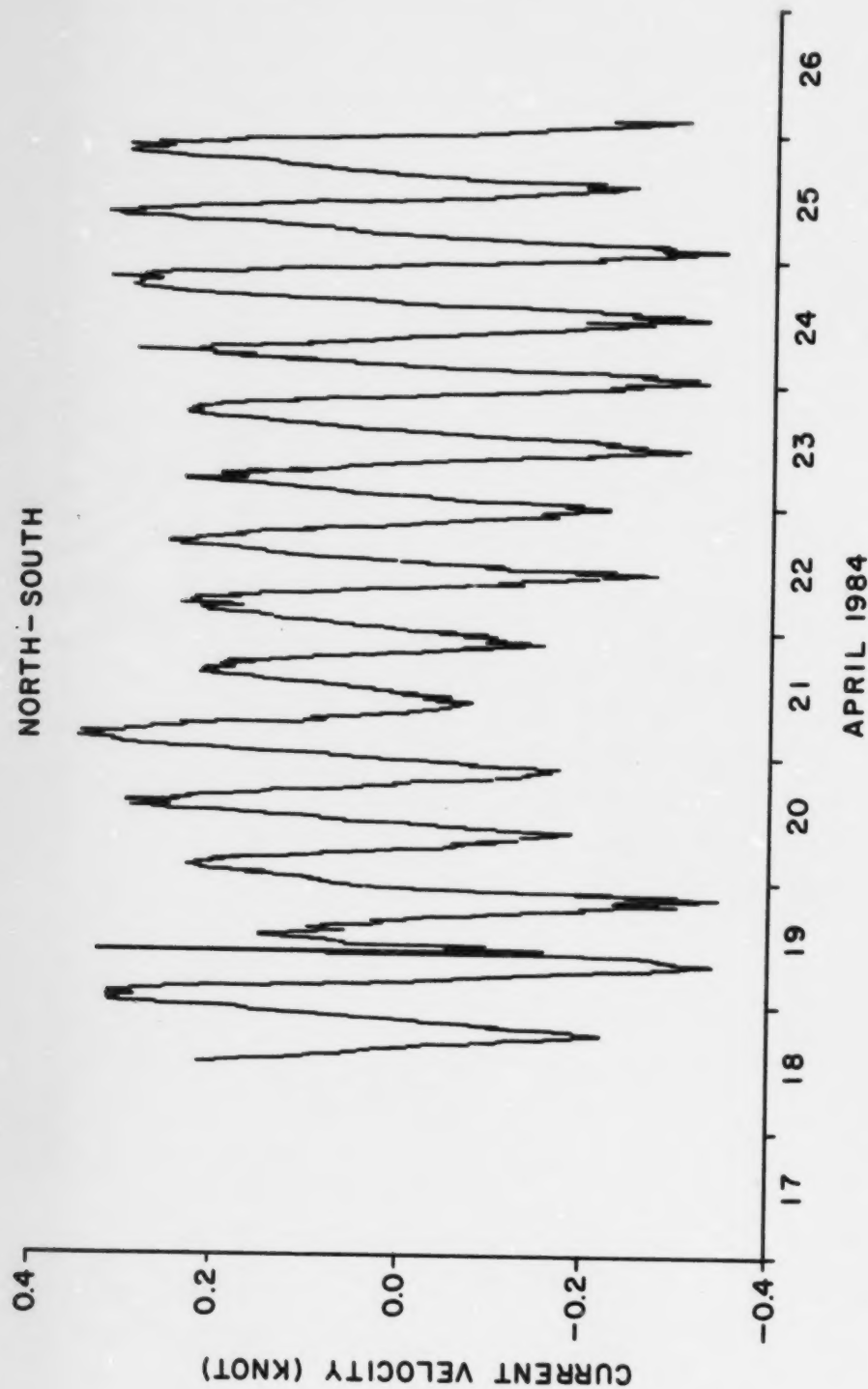


Figure 22. North-south current velocity at 1 m above bottom during the 17 April 1984 deployment of the in situ instrumentation array.

LIS and weather-induced surface activity. Complete descriptions of current data collected through the use of the in situ instrument array may be found in Bohlen and Winnick (1984a,b, 1986).

Temperature

Surface and near-bottom water temperatures measured during ERLN cruises to the FVP study area are presented in Figure 23. Multiple data points on given dates correspond to repeated measurements at single stations as well as those obtained at different stations over the course of a single day. Temperatures ranged from approximately 0°C in late winter to near 25°C in late summer. Similar patterns have been reported for CLIS by Riley (1956), Rhoads and Boyer (1982), and others.

Smoothed curves in Figure 23 represent the results of harmonic regression analyses (for all stations combined), which accounted for 98.0 and 98.4% of the total variation in surface and bottom data, respectively. Variability observed during single cruise dates was the result of both solar insolation (surface waters) and tidally-induced horizontal movement of inhomogeneous water masses through the study area (surface and bottom waters). The influence of surface heating and atmospheric cooling is apparent in Figure 24, which shows a series of five vertical, temperature profiles taken at CTR over the course of a single cruise on 22 April 1983. Surface temperature increased approximately 2°C, reaching a maximum at 17:00, then cooled as the day progressed. Variation in surface temperature associated with the horizontal movement of water masses was apparent in images obtained from the AVHRR sensor on the NOAA 7 satellite taken during spring of 1983. These showed differences of several tenths of a degree across the Sound. Somewhat larger variation in surface temperatures (up to 6°C) were found by Reid et al. (1979) during April 1973 cruises. Thermal

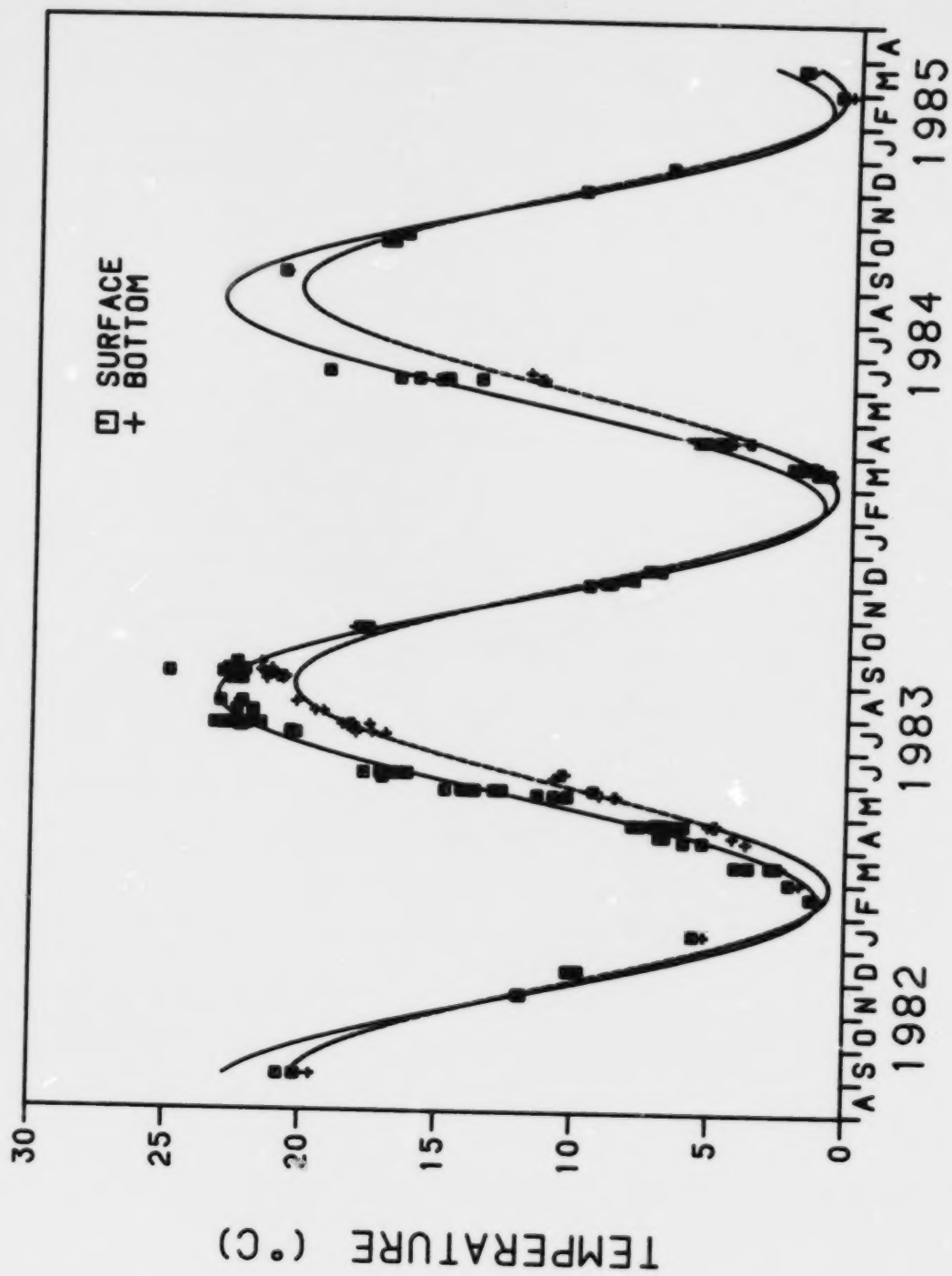


Figure 23. Surface and near-bottom water temperatures at the FVP study site.

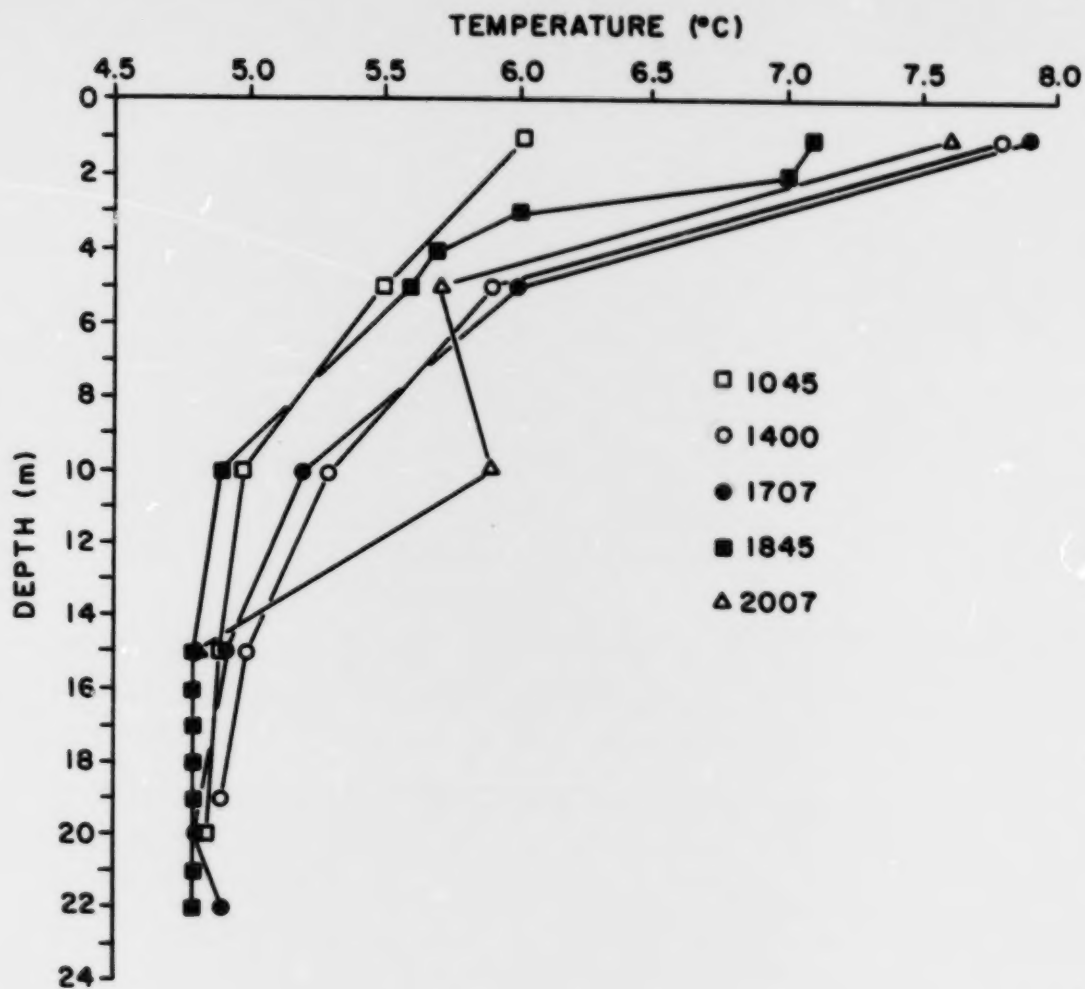


Figure 24. Vertical temperature profiles measured at CTR on 22 April 1983.

variability associated with water movement is also evident in the record of bottom water temperature obtained by the in situ instrument array, an example of which is given in Figure 25 from the 18 April 1983 deployment. Tidally-induced current flow during this deployment yielded variation on the order of 0.2-0.5°C over a 6 hour period. Near-bottom temperatures in Figure 24 reflect similar variation over the course of the cruise.

Thermal stratification in CLIS began in March-April in all years studied (Figure 23). Differences between surface and bottom temperatures reached a maximum in July and August, at which time surface waters were up to 4°C warmer than bottom waters. This structure broke down fairly rapidly in early fall, such that by September-October vertical variability was less than that associated with tidal movement of water masses. Thereafter, water temperatures remained fairly uniform with depth until the following spring.

Salinity

Salinities measured in surface waters and at 1 m above bottom during ERLN cruises to the FVP study area are shown in Figure 26. Multiple data points on given dates correspond to repeated measurements at single stations as well as to those obtained at different stations over the course of a single cruise. Salinity displayed seasonal patterns which appear strongly influenced by freshwater input to the Sound, as represented by the stream flow data presented earlier (see Figure 20). Lowest salinities in both surface and bottom waters corresponded to the period of maximum runoff during the spring freshet, while higher salinities were recorded during the drier seasons of each year. Stream runoff in December 1983 and June 1984 was unusually high, apparently resulting in the more prolonged salinity depression in 1984, as compared to 1983.

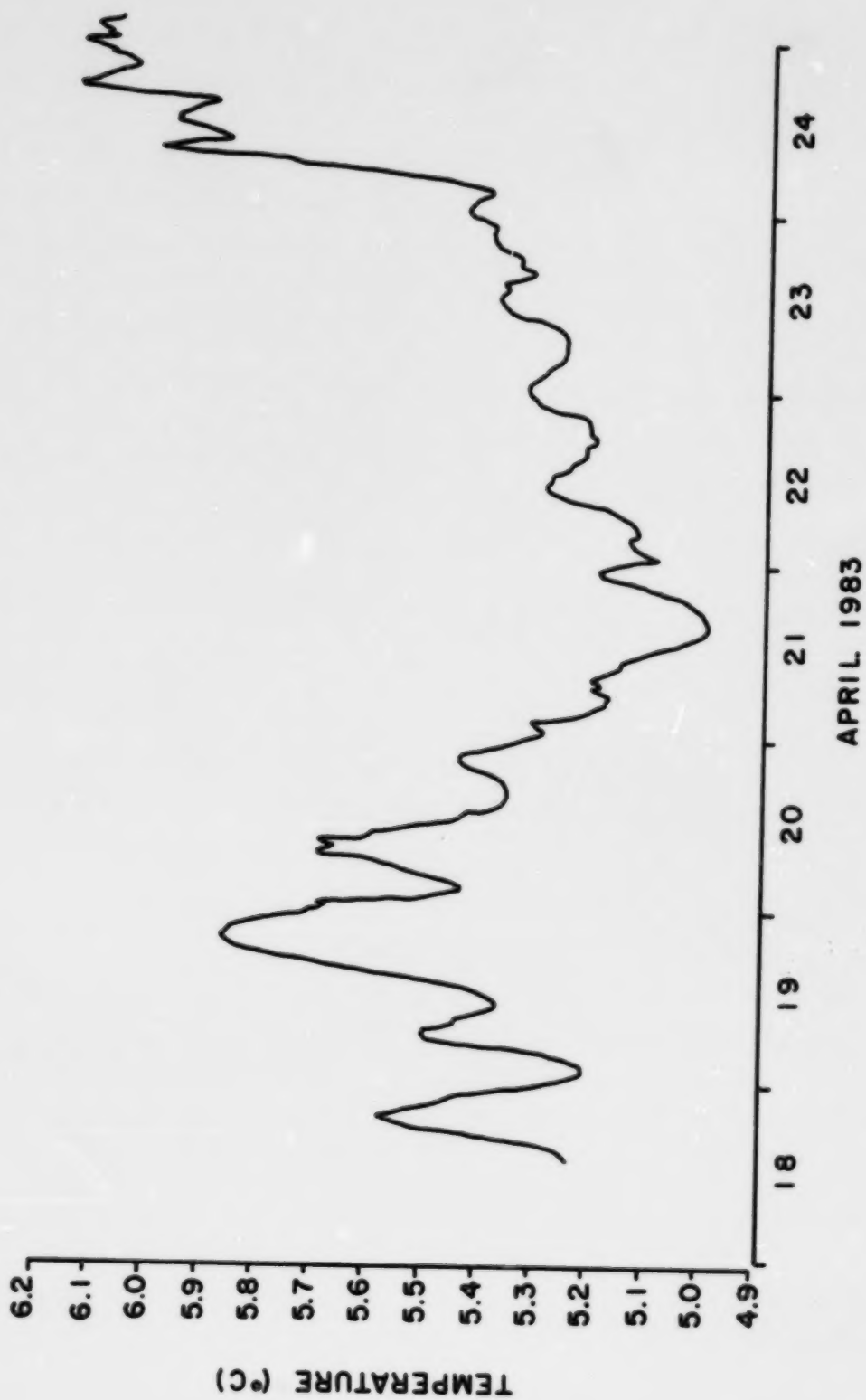


Figure 25. Water temperature at 1 m above bottom during the 18 April 1983 deployment of the in situ instrumentation array.

However, the relative paucity of data in 1984 renders this conclusion somewhat tenuous. Surface salinities (which ranged from 23 to 29 ppt) were affected by runoff to a larger extent than were those of bottom waters (which ranged from 24 to 29 ppt) in a manner consistent with Riley's (1956) description of estuarine circulation in the Sound: denser shelf water with more or less uniform salinity enters the Sound through The Race along the bottom, mixes with less dense freshwater entering from river systems displaying seasonally fluctuating discharge rates, and leaves the Sound near the surface. Due to this pattern of circulation and surface runoff, surface salinities were generally lower than those measured near-bottom.

Vertical salinity profiles obtained during the 22 April 1983 cruise are shown in Figure 27. These profiles were collected between 10:45 and 20:07 at the CTR station. A pycnocline occurred near the 10 m depth, such that a difference of up to 3 ppt existed between surface and bottom waters at various times of the day. The diel variation present in Figures 26 and 27 resulted in large part from the horizontal movement of inhomogeneous water masses through the study area. Figure 28 shows salinity measurements obtained from the 18 April 1983 deployment of the in situ array. As was seen for temperature (Figure 25), salinity fluctuations coincident with tidal flow were apparent. Surface salinities were affected to a greater extent (Figure 27) by riverine inflow. Larger scale variability seen in Figure 28 may have resulted from the passage of an atmospheric front.

Dissolved Oxygen

Dissolved oxygen (DO) values measured during ERLN cruises to the FVP study area in 1983 at 1 m above bottom are presented in Figure 29, along with calculated saturation levels based upon measured temperature and salinity. DO values were at or near

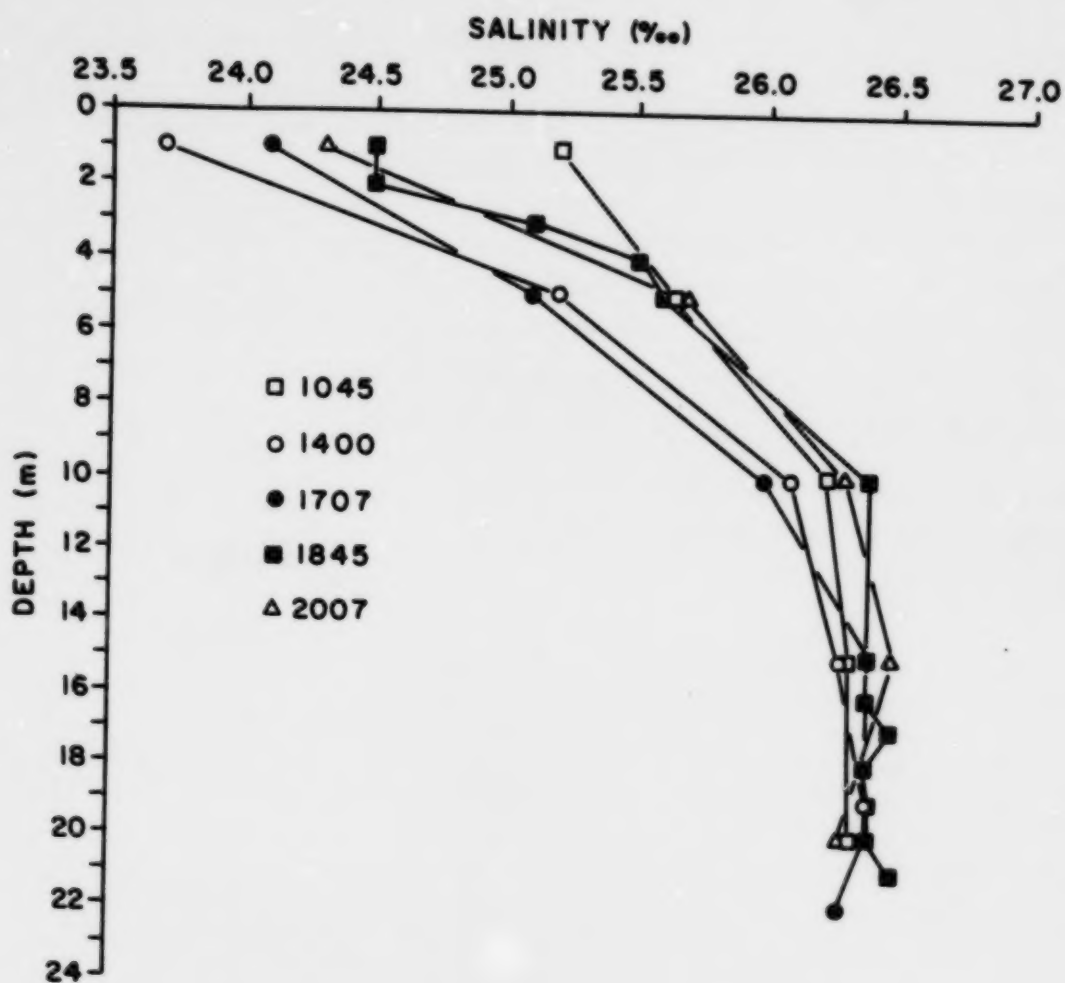


Figure 27. Vertical salinity profiles measured at CTR on 22 April 1983.

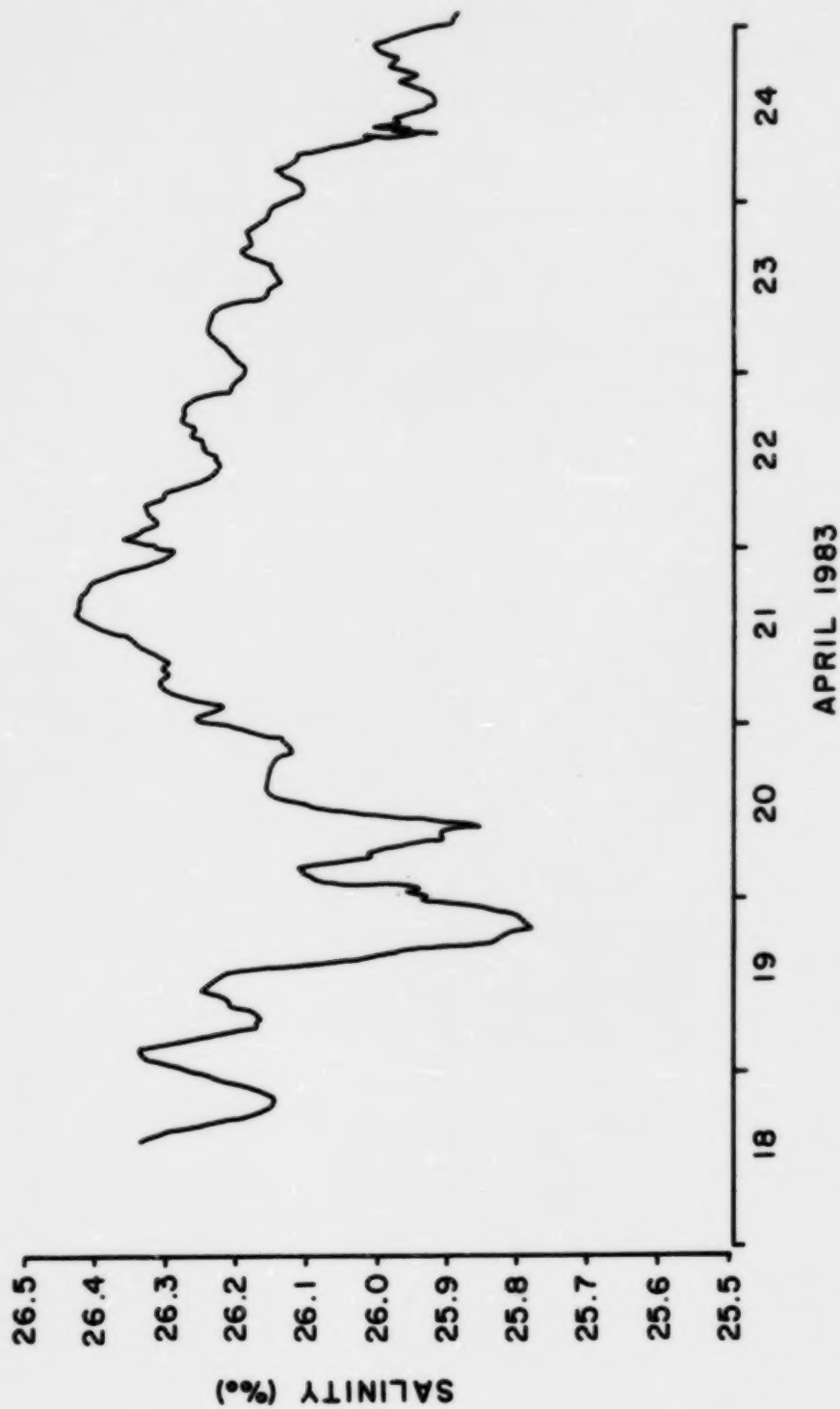


Figure 28. Salinity at 1 m above bottom during the 18 April 1983 deployment of the in situ instrumentation array.

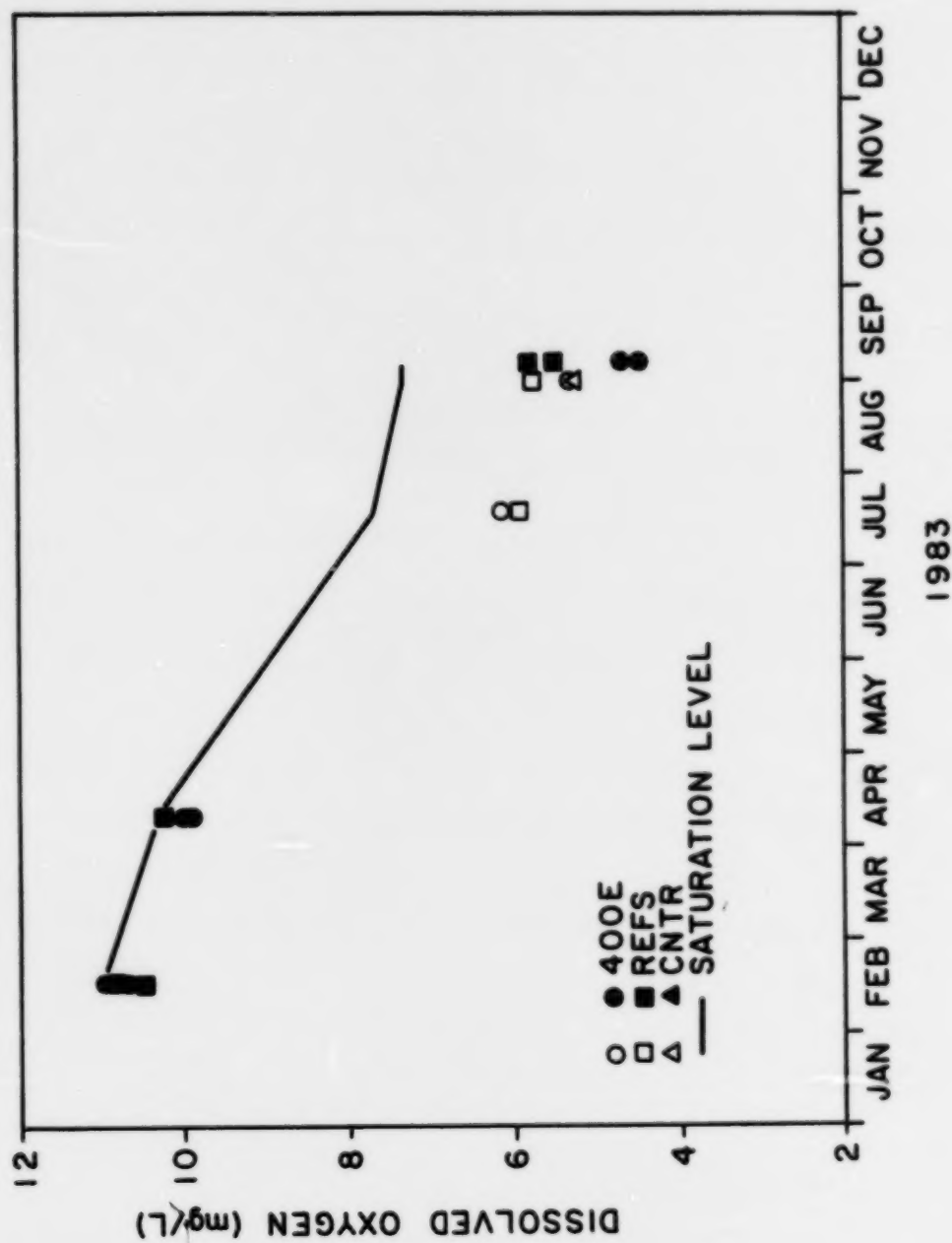


Figure 29. Near-bottom dissolved oxygen in 1983.

saturation levels in February and April prior to the disposal operation, but were somewhat depressed following disposal. There is strong reason to believe that this depression resulted from influences other than the presence of dredged material, as DO levels were also depressed at the reference station (REFS) during this later period (Figure 29). In no case, however, did DO levels approach hypoxic conditions, as the lowest values measured at any station were greater than 4 mg/l. Although the causes of the observed oxygen depression are unknown, possible explanations include increased benthic respiration and microbial decomposition rates resulting from the increased water temperature during this period (see Figure 23) in conjunction with the effects of the seasonal pycnocline. No further DO measurements were made following the cruises in early September 1983.

Figure 30 shows DO profiles obtained during a cruise on 2 September 1983. The pycnocline on this date occurred at a depth of approximately 10 m, and acted as a barrier to vertical exchange of oxygen. Above this depth, waters were near saturation due to atmospheric input of oxygen and turbulent mixing, in addition to the release of oxygen from primary producers. DO levels below the pycnocline were relatively lower due to respiration, decomposition, and the isolation from surface waters.

The near-bottom data in Figure 30 appear to indicate an increasing trend in DO with distance from CTR, with oxygen concentrations highest at REFS. However, differences between stations at this depth were well within the variability displayed in the surface waters, where the apparent trend did not exist. Thus there is little indication that the presence of dredged material affected DO concentrations near the mound.

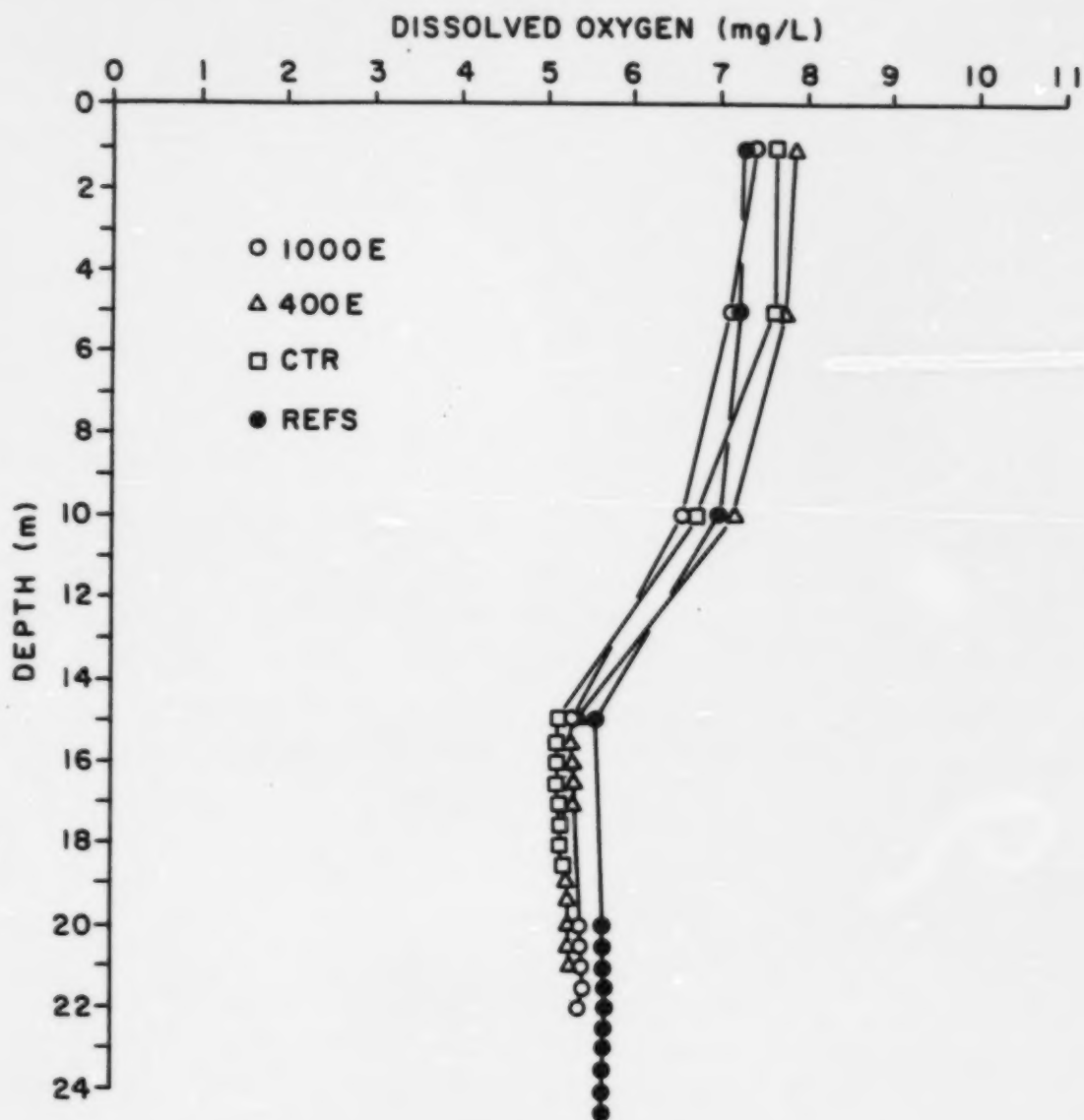


Figure 30. Vertical dissolved oxygen profiles measured at CTR on 2 September 1983.

Turbidity and Suspended Solids

The normal range of suspended solids concentrations encountered at 1 m above bottom at the FVP study area was approximately 5-25 mg/l. The highest concentrations measured at this depth during ERLN cruises approached 35 mg/l during December 1983, a period of relatively high storm activity and runoff (see Figure 20). These values, although in close agreement with those reported by others (e.g. Rhoads and Boyer, 1982), are considerably lower than concentrations measured in many freshwater systems (e.g. Paul *et al.*, 1982; McIlroy, 1984). Surface concentrations were generally lower than those obtained near-bottom.

The relationships between light transmission and suspended solids concentration obtained during ERLN cruises are presented in Figure 31. Curves correspond to least-squares regressions calculated for each cruise separately. The results of these analyses are presented in Table 16. Although there seemed to be a seasonal effect in the relationship of these two parameters (*sensu* Baker and Lavelle, 1984), apparent differences were confounded by the use of transmissometers with different path lengths. Variation between dates (other than that due to the transmissometers) presumably resulted from differences in the composition and size distributions of suspended particles (Baker and Lavelle, 1984).

Examples of vertical profiles of light transmission are given in Figure 32. These profiles were taken at CTR over the course of a single cruise on 22 April 1983. Minimum turbidity occurred at a depth of approximately 7 to 10 m in each profile, corresponding to the depth of the pycnocline on that date (see Figures 24 and 27). Light transmission below the pycnocline decreased with increasing depth due to the resuspension and mixing of fine grained bottom sediment by tidal action and

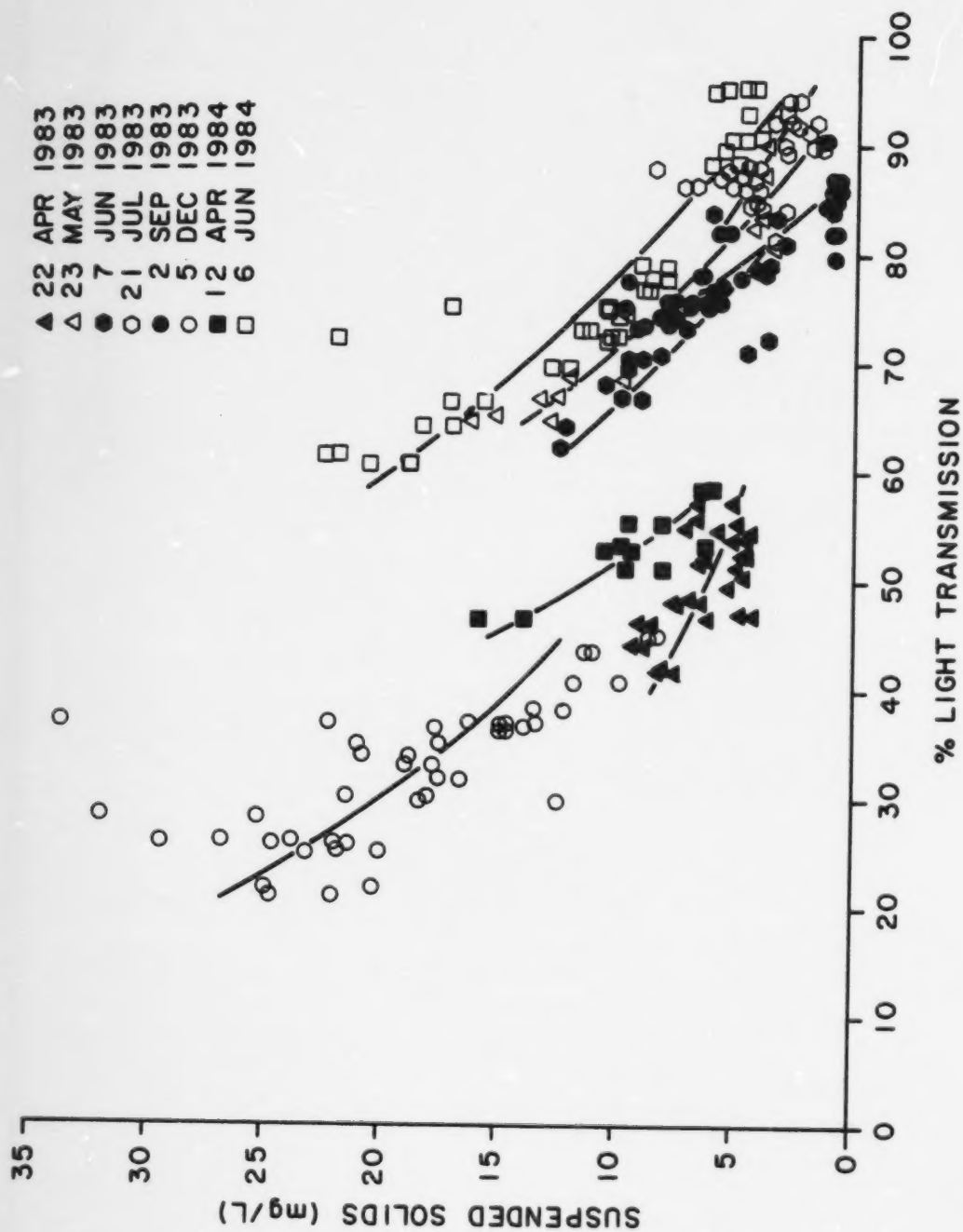


Figure 31. Relationships between light transmission and suspended solids concentration during ERLN cruises.

TABLE 16. PARAMETER ESTIMATES FROM LEAST-SQUARES REGRESSIONS OF SUSPENDED SOLIDS CONCENTRATION (S) AGAINST NATURAL LOGARITHM OF % LIGHT TRANSMISSION (T) USED IN THE CALIBRATION OF VERTICAL LIGHT TRANSMISSION PROFILES

Date	a*	b*	df	F	R ²
22 Apr 1983	-10.30	46.45	29	17.98	0.38
23 May 1983	-35.70	162.48	16	135.19	0.89
7 Jun 1983	-31.54	142.29	20	68.71	0.77
21 Jul 1983	-25.77	119.50	44	19.16	0.30
2 Sep 1983	-47.02	210.42	54	212.35	0.80
5 Dec 1983	-18.85	83.85	44	38.24	0.46
21 Apr 1984	-35.04	148.34	10	29.54	0.75
6 Jun 1984	-35.87	166.35	40	185.91	0.82

* a = slope
b = intercept

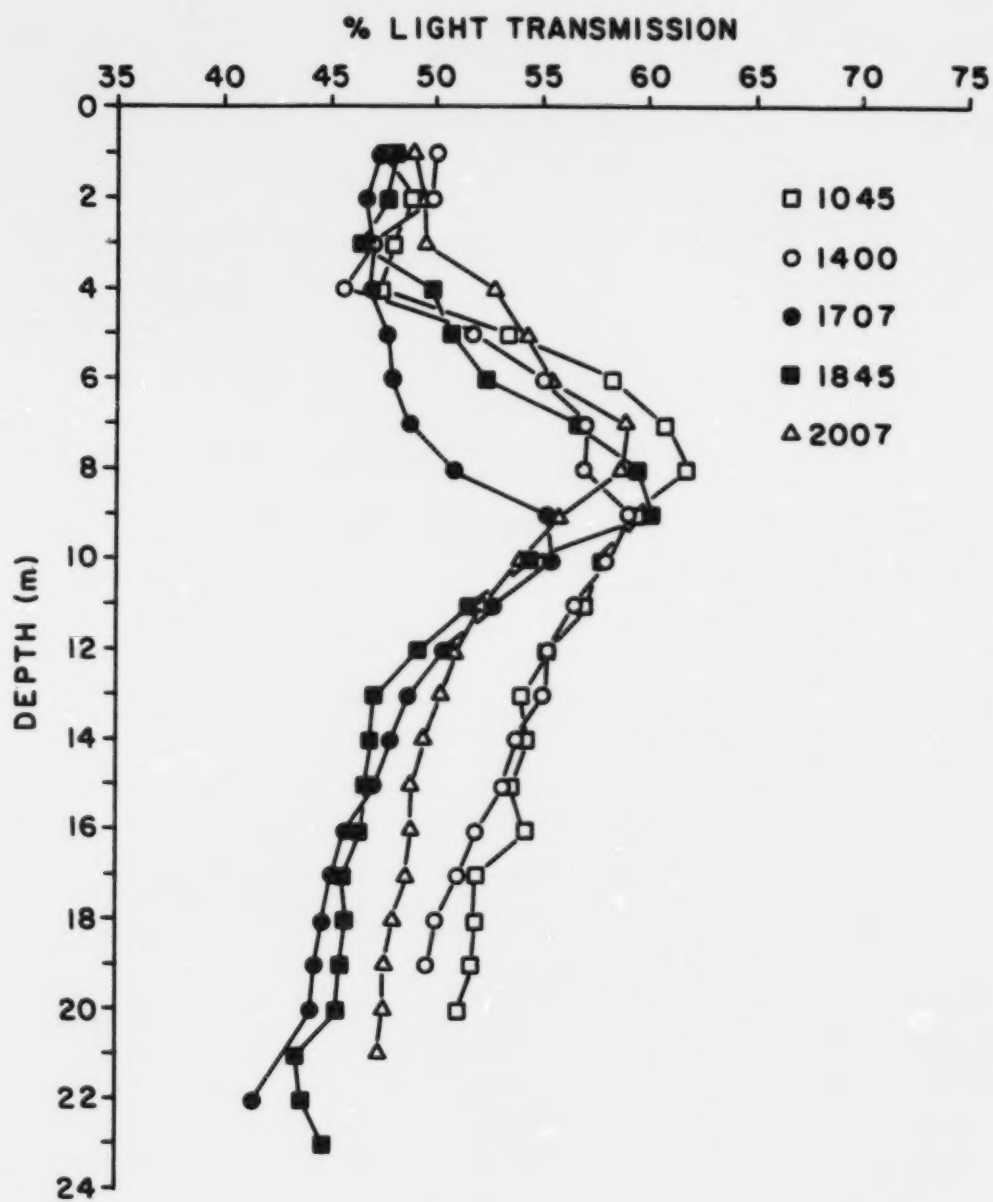


Figure 32. Vertical light transmission profiles measured at CTR on 22 April 1983.

biological activity (see for example, Davis et al., in prep.). Above the pycnocline, transmission again decreased, presumably due to primary production at the surface. Vertical profiles similar (though less pronounced) to these were measured by Rhoads and Boyer (1982). Variation in near-bottom light transmission resulted chiefly from changes in tidal current speed. Diel variation, albeit relatively small, was also seen in surface waters. Possible explanations for this variation include changes in plankton vertical distributions as well as horizontal movement of inhomogeneous water masses.

Suspended solids concentrations estimated at 1 m above bottom from light transmission data obtained during the 26 April 1983 deployment of the DAISY in situ array at 1000E can be seen in Figure 33. Background concentrations averaged approximately 9 mg/l during this deployment. The influence of tidal currents is apparent in variation of solids concentration occurring over periods of 6 hr. Superimposed on this pattern were brief excursions from background levels of up to 45 mg/l. Because both the frequency and magnitude of these peaks were reduced during deployments when CLIS disposal operations were not taking place, one cause of their occurrence may have been the passage of disposal plumes past the 1000E station. This indicates that dredged material may have been transported (and deposited) outside of the immediate disposal area. Other potential causes of these peaks include surface wave conditions, activities of the local commercial fisheries, biologically mediated resuspension events, or temporary interference of the transmissometer light path by motile, epibenthic organisms and floating macroalgal thalli.

Suspended solids concentrations from the sediment-water interface to 1 m above bottom were determined by the OSU acoustic profilometer. Results indicate that concentrations near the sediment-water interface could be up to 30-50 times higher

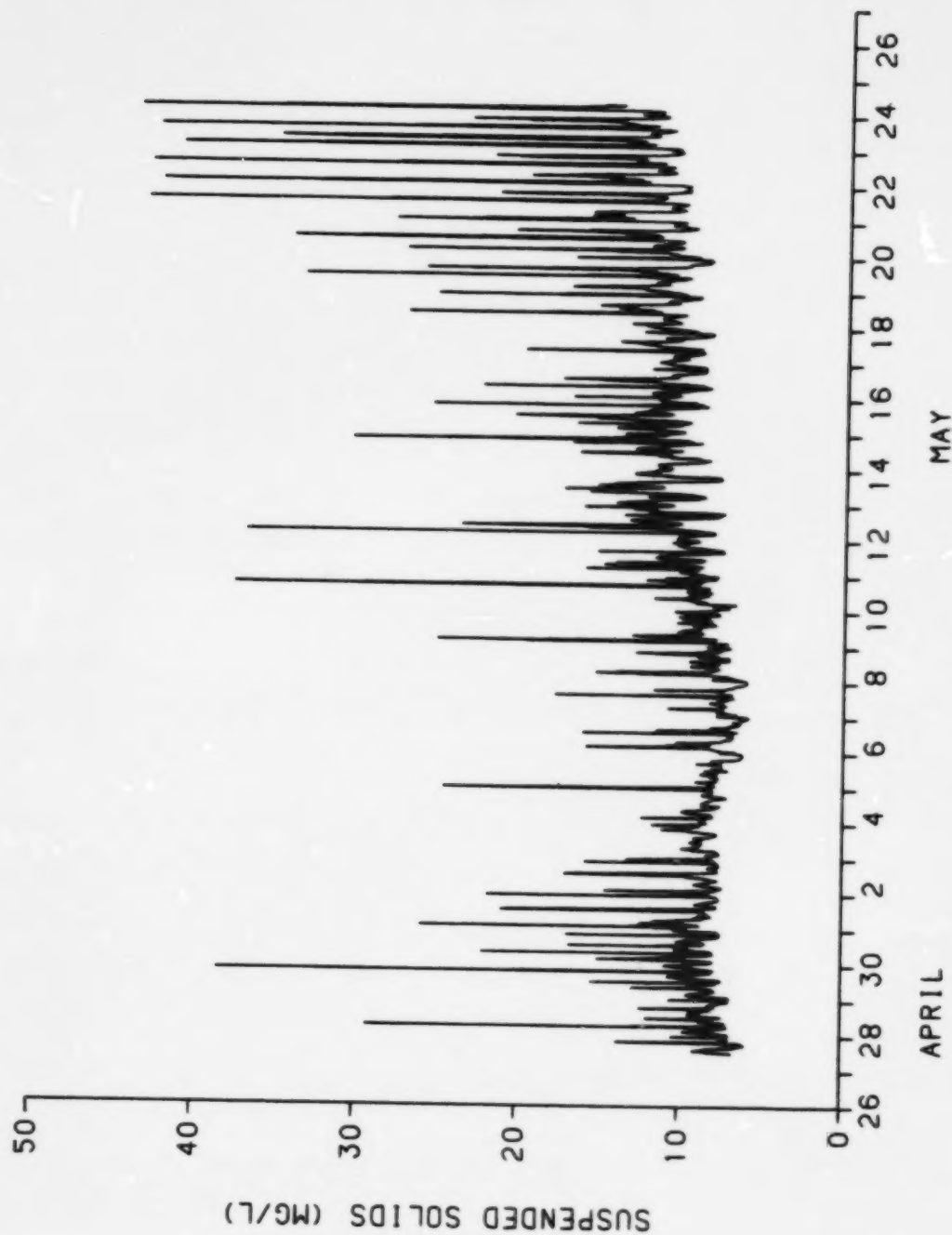


Figure 33. Suspended solids concentrations at 1 m above bottom during the 26 April 1983 deployment of the in situ instrumentation array.

than those at 1 m above bottom. Figure 34 shows a depth profile of suspended solids concentration relative to that determined at 0.5 m above bottom (mid-depth) during the 13 June 1984 deployment of the profilometer. Concentrations increased in an exponential fashion with depth from a minimum of 0.3 times that at mid-depth near 1 meter above bottom to a maximum of 7 times that at mid-depth immediately above the bottom due to resuspension of fine material by the turbulent activity of currents. The occurrence of this 'fluff layer' suggests that organisms living on or near the sediment-water interface can be subjected to much higher levels of particulate-associated contaminants, as compared to exposure levels at 1 m above bottom (see chemistry data below).

Sediment Physical Characteristics

FVP Study Reach Sediments--

Sediments collected in the FVP study reach prior to dredging consisted of organic, clayey silt with traces of fine sand and larger gravel. Physical characteristics of these sediments are summarized in Table 17. Large variation was observed among samples collected from different regions of the study reach, reflecting patterns of surface runoff from the surrounding areas. For instance, median grain size ranged over an order of magnitude, from a low of 0.001 mm up to 0.024 mm. Grain sizes denoting the 75th quartile (i.e. the size below which 75% of the particles fell) displayed similar variation (Table 17). Percent water content was relatively more uniform, ranging from 61.1 to 77.3%.

The water content of BRH material used in laboratory effects studies was 69.0%, similar to that observed for study reach sediments. However, analyses performed by the University of Connecticut suggested a size distribution shifted towards larger particle sizes. The median particle size of samples from

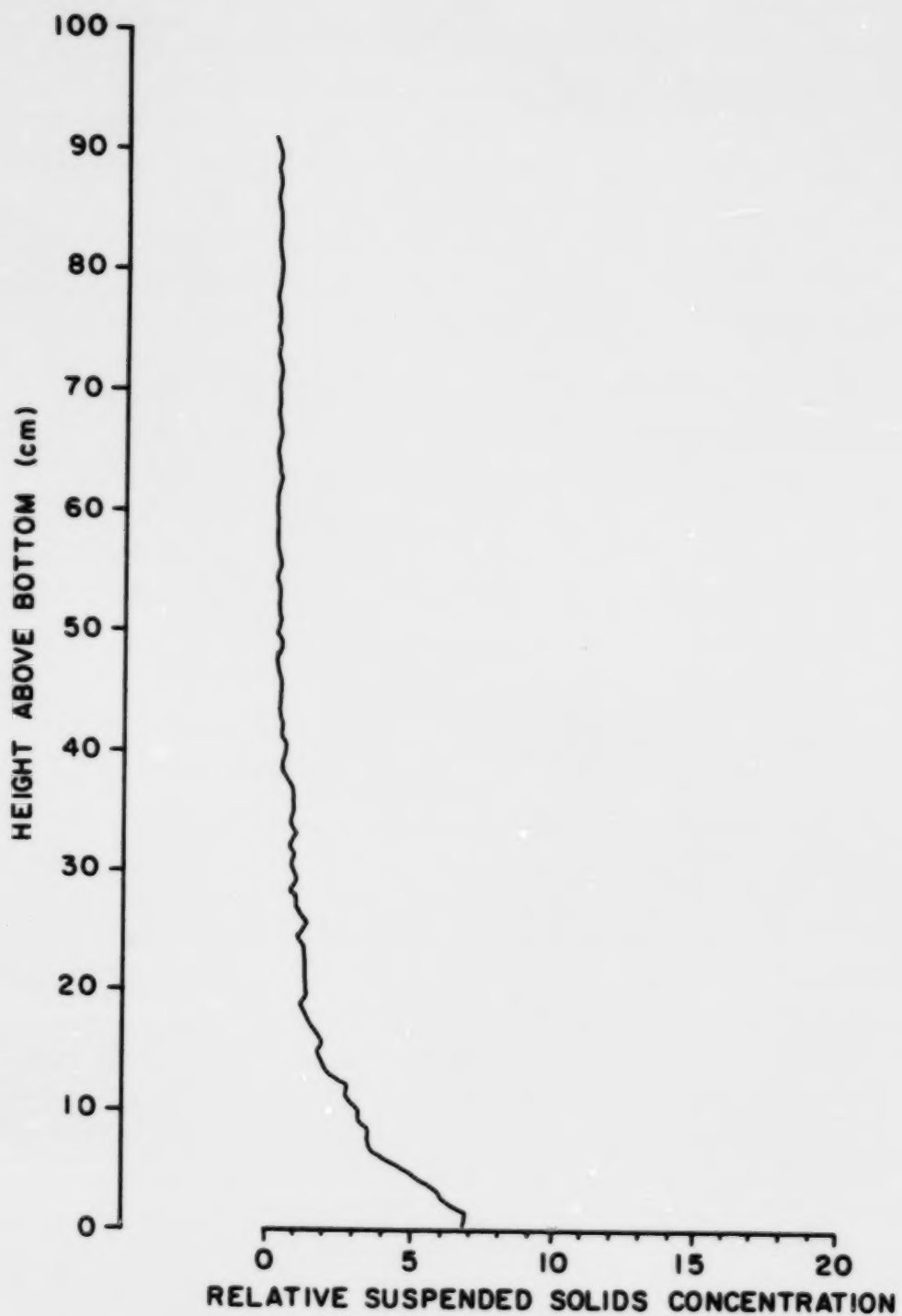


Figure 34. Relative suspended solids concentrations between the sediment-water interface and 1 m above bottom.

TABLE 17. PHYSICAL CHARACTERISTICS OF FVP STUDY REACH SEDIMENTS

	% Water content	Median grain size (mm)	25th quantile	75th quantile	% Fines (<0.063 mm)
Median	72.6*	0.011	0.006	0.021	93.0
Maximum	77.3	0.024	0.009	0.110	98.0
Minimum	61.1	0.001	0.004	0.013	65.0
Range	16.2	0.023	0.006	0.097	33.0

* n=25

two barrels was 0.030 mm, compared with the 0.011 mm measured in the study reach. In addition, the proportion of particles less than 0.063 mm in the barrel material was 81.0%, as opposed to the 93.0% found in the reach. Although some fine particulates may have been lost during collection of the barrel material, these discrepancies were probably due in large part to differences in analytical methods used between NED and the University of Connecticut.

Field Sediments--

Sound-wide patterns of sediment particle size may be seen in Figure 35 (modified from Bohlen and Bedford, 1983). The hatched rectangle in this figure indicates the location of the CLIS Disposal Site. These patterns reflect areas of long-term sediment erosion and deposition resulting, in large part, from the tidally-induced current structure of the Sound described earlier. Bottom sediments near the relatively shallow Race are dominated by coarse-grained material due to high current speeds and tidal scouring. The CLIS basin is dominated by fine-grained sediments resulting from sediment deposition. Distributions of particle sizes in surficial sediments at the FVP disposal site were fairly uniform prior to disposal. Histograms of size distributions measured at CTR, 1000E, and REFS on 4 March 1983 are given in Figure 36. Sediments from all stations were dominated by particles less than 3.9 μm , and the percentage of fine particles ranged from 90.4 to 93.0%. Mattituck Sill separates this central basin from the eastern portion of the Sound and the influence of The Race. Similar descriptions of grain size distribution in the Sound are given by Reid et al. (1979) and Hunt (1979).

Between 26 April and 14 May 1983, approximately 55,000 cubic meters of BRH material were disposed by point dumping at the designated FVP site (41° 9.39' N, 72° 51.75' W). Bathymetry surveys were carried out as part of the DAMOS project

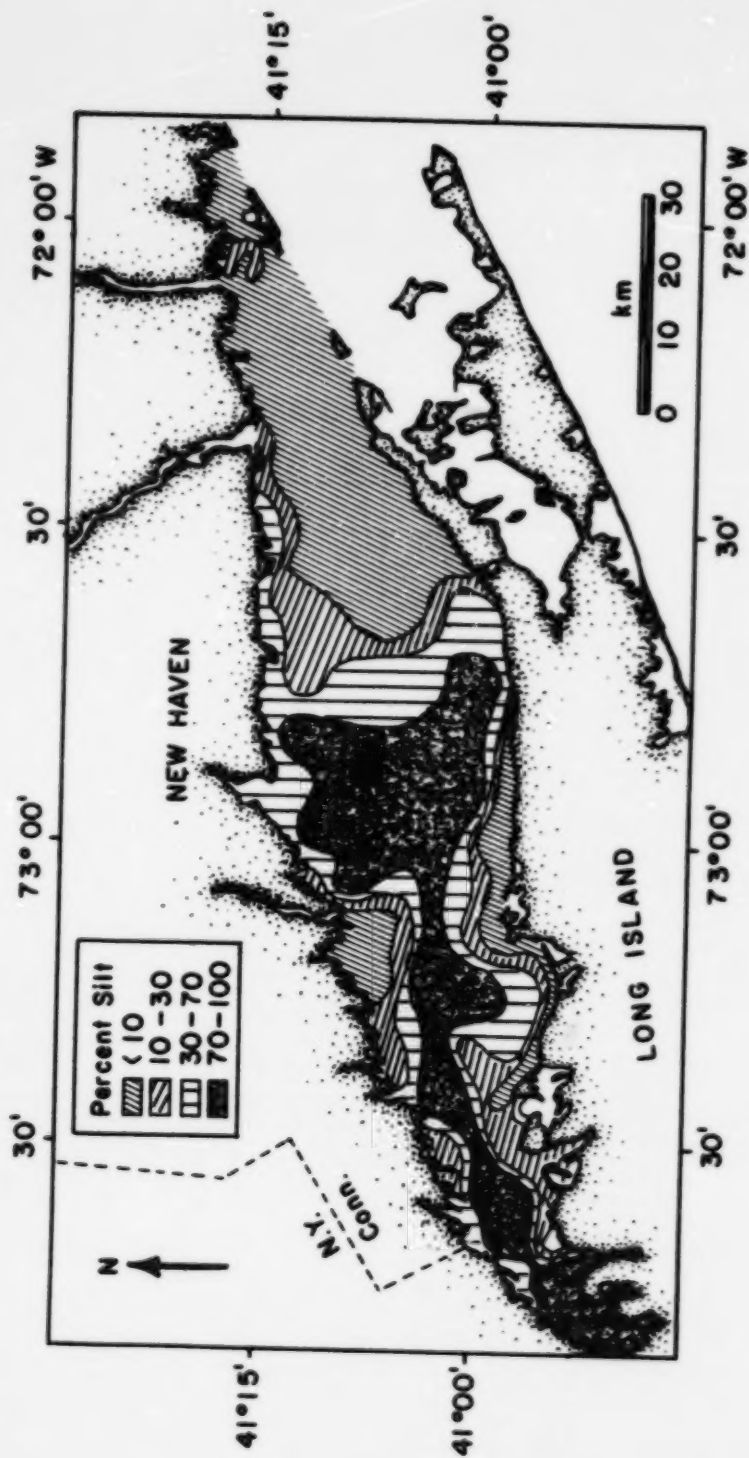


Figure 35. Bottom sediments in Long Island Sound.

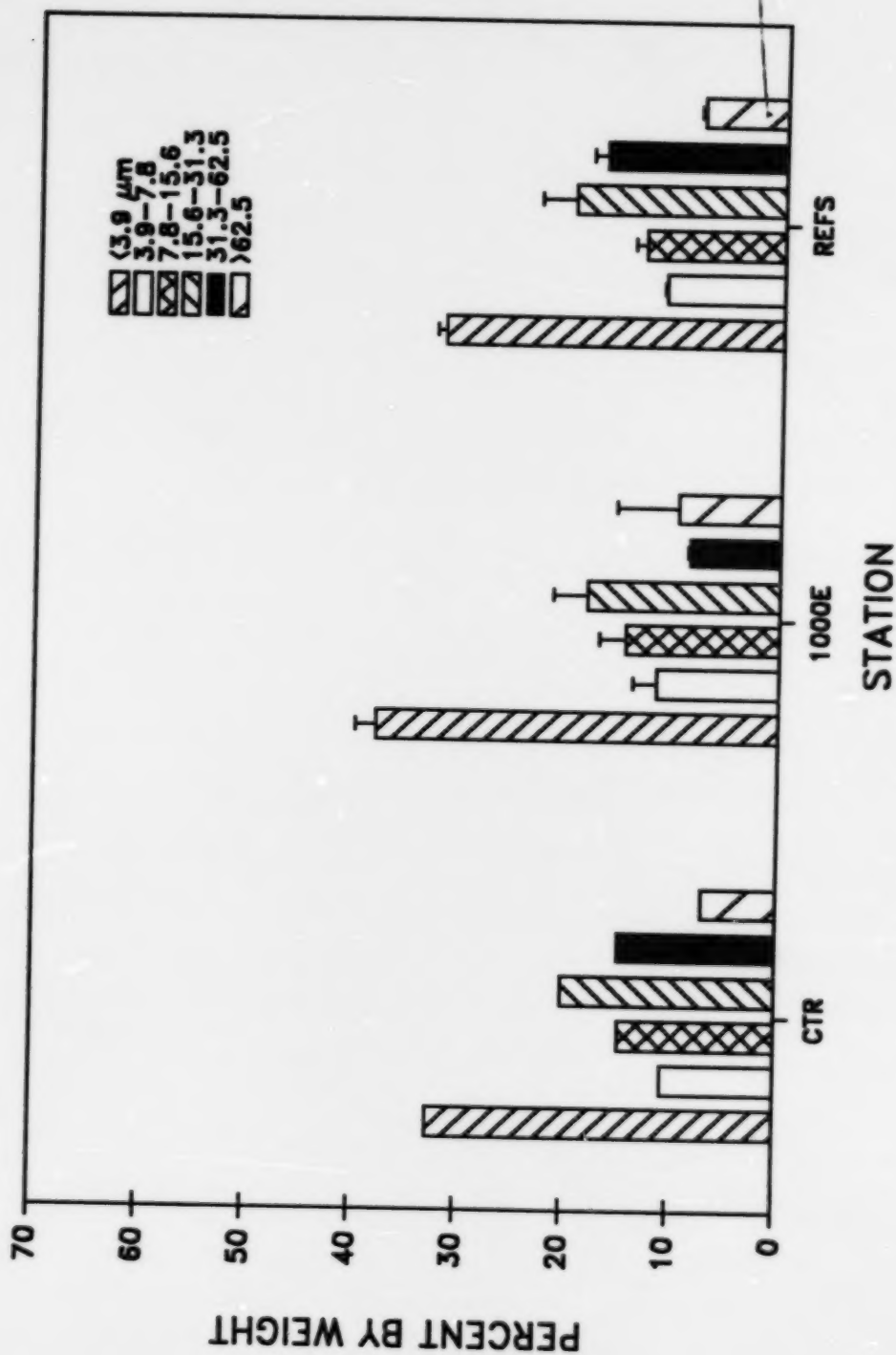


Figure 36. Particle size distributions of predisposal biological effects station sediments.

by Science Applications International Corporation (SAIC) in conjunction with REMOTS® surveys to monitor changes in depth of the disposal mound following the disposal operation (see Scott et al. (1984) for complete details). A maximum dredged material thickness of 1.8 m was observed at CTR immediately following cessation of disposal (see Figure 37). The thickness of the deposit was reduced to several cm at a distance of 100-150 m from CTR, with a thin veneer extending to approximately 400 m. A survey carried out in June 1983 indicated a decrease in thickness of 20-40 cm at CTR, chiefly due to consolidation of material in the disposal mound. However, some of this decrease in thickness may have been due to a net flux of material off the mound.

Particle size distributions of surficial sediments collected at the biological effects stations on 3 June 1983 are displayed in Figure 38. Sediments at CTR and 200E appeared to have been enriched with coarse-grained particles to the extent that particles greater than 0.063 mm dominated the sediments at these two sites. Evidence of enrichment was also observed at 400E, though to a lesser extent than seen at CTR and 200E. Although size distributions gradually sifted towards smaller particle sizes over time, coarse-grained particles continued to dominate the sediments of mound station stations through early September 1983. Sediments at 1000E and REFS were unaffected by the disposal operation, and displayed size distributions similar to those obtained prior to disposal. Particle distributions measured at REFS throughout the summer following disposal are shown in Figure 39.

The differences observed in granulometry between BRH barrel material and mound surface sediments suggest a loss of fine sediments during the dredging, barging, and disposal operations, loss of material immediately subsequent to disposal as a result of tidal scouring, and the addition of coarse particles to the

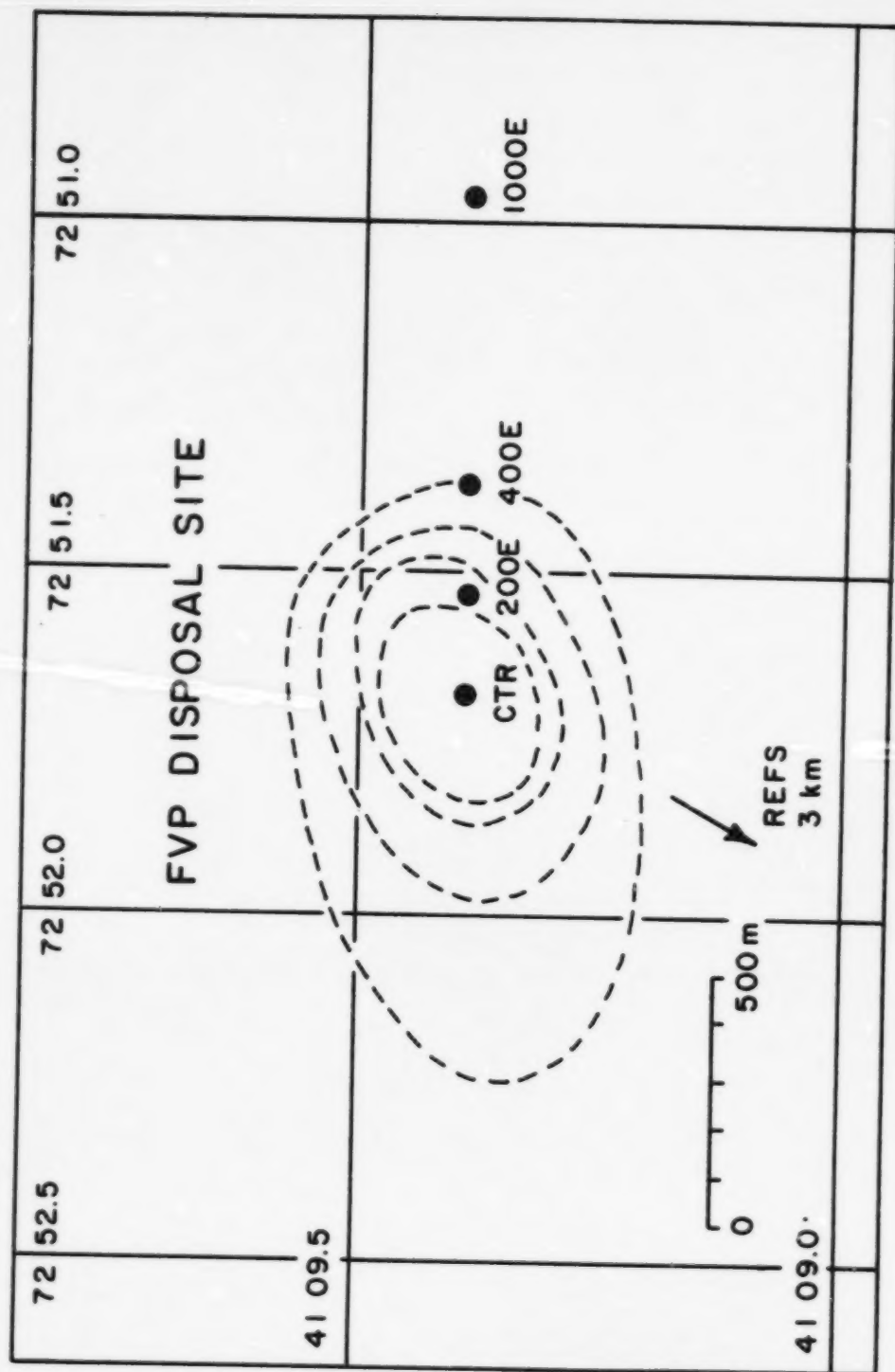


Figure 37. Topography of FVP disposal site in June 1983.

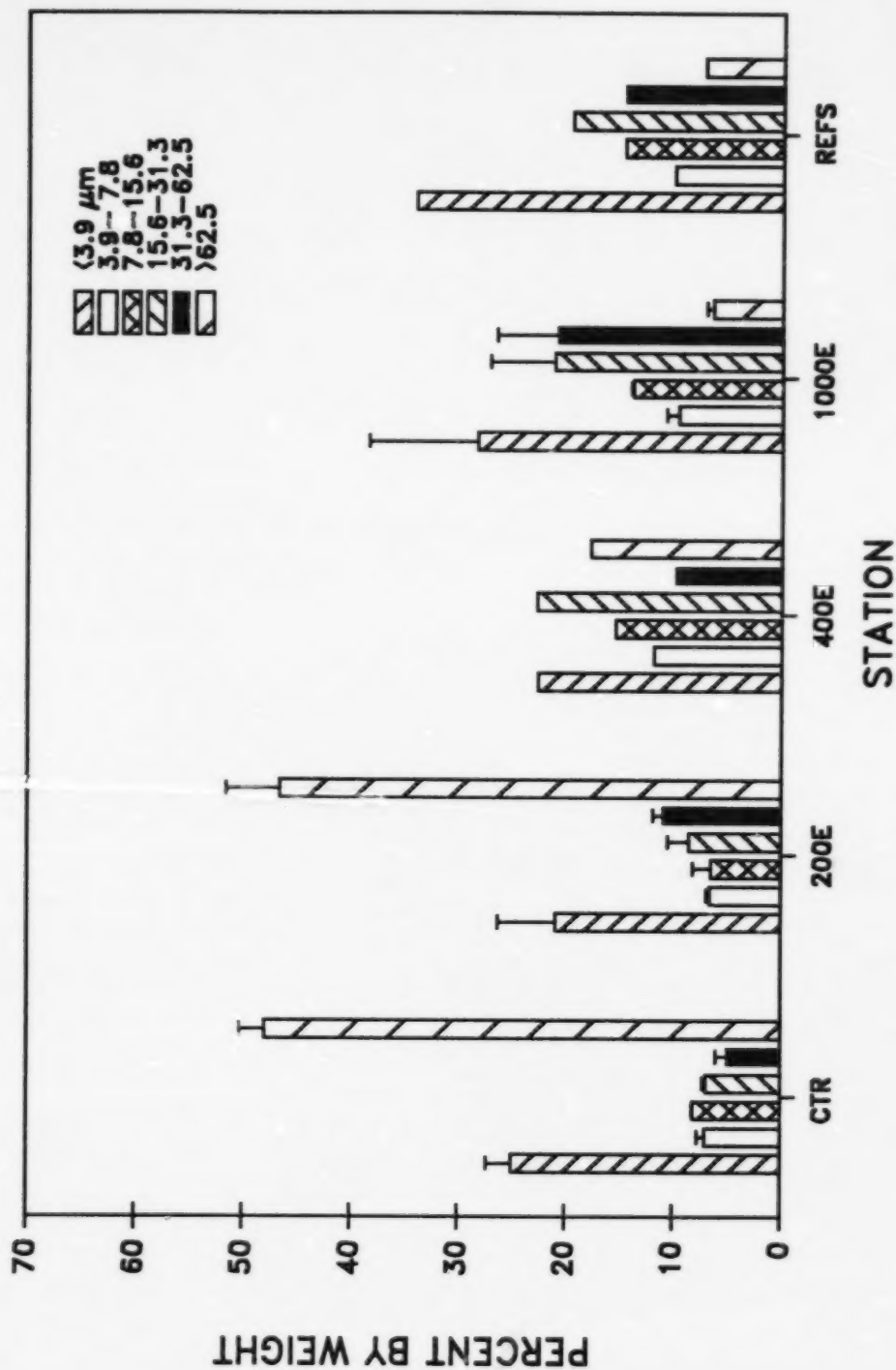


Figure 38. Particle size distributions of postdisposal biological effects station sediments.

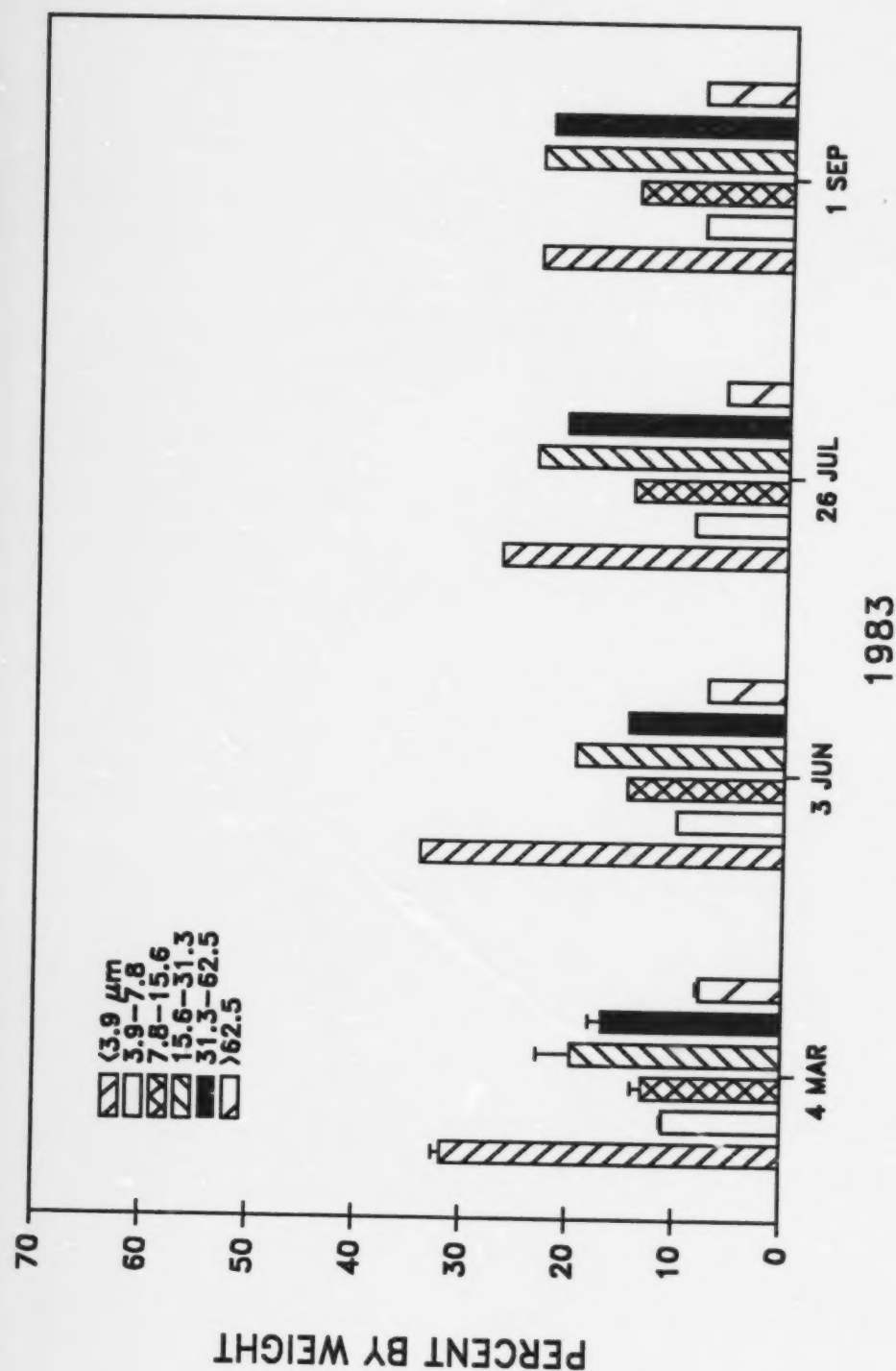


Figure 39. Particle size distributions of REFS sediments over time.

material (in the form of bagged sand) eventually released from barges. The relative contributions of these processes are unclear. Retention of material in the water column during disposal was detected as an increase in suspended solids excursions measured by in situ instrumentation (see Figure 33). However, loss of fine particles to the water column during dredging and disposal would have affected the relative distribution among size classes below 0.063 mm, as a result of the positive relationship between particle size and settling velocity. Whereas the postdisposal distribution observed at 200E suggests such a loss, differences in the relative proportions in the fine size classes at CTR were hard to detect. In addition, tidal scouring might be expected to affect size distributions in a similar fashion. Finally, much of the shift in particle sizes could be attributed to enrichment by sand, as sand bags were used during the barging operation to aid in the retention of dredged material in scows (K.J. Scott, SAIC, personal communication). Gravity cores taken on the mound in 20 March 1985 appeared marbled, with regions of fine sediments sandwiched between regions of (apparently) clean sand and relic clay clasts. This vertical variation resulted from the addition of sand as well as from differences between barge loads of dredged sediments.

Further evidence of loss of fine material and enrichment by sand in CTR sediments can be found in a comparison of bulk sediments collected from CTR on 18 October 1983 with those from ERLN barrels effects experiments (Figure 40), as determined by the Malvern particle sizer. Approximately 50% of CTR material was greater than 0.26 mm, compared with approximately 8% in barrel material. When particles greater than 0.26 mm were removed from samples by sieving, particle size distributions were more similar (Figure 41), although the median grain size of barrel sediments was still somewhat smaller.

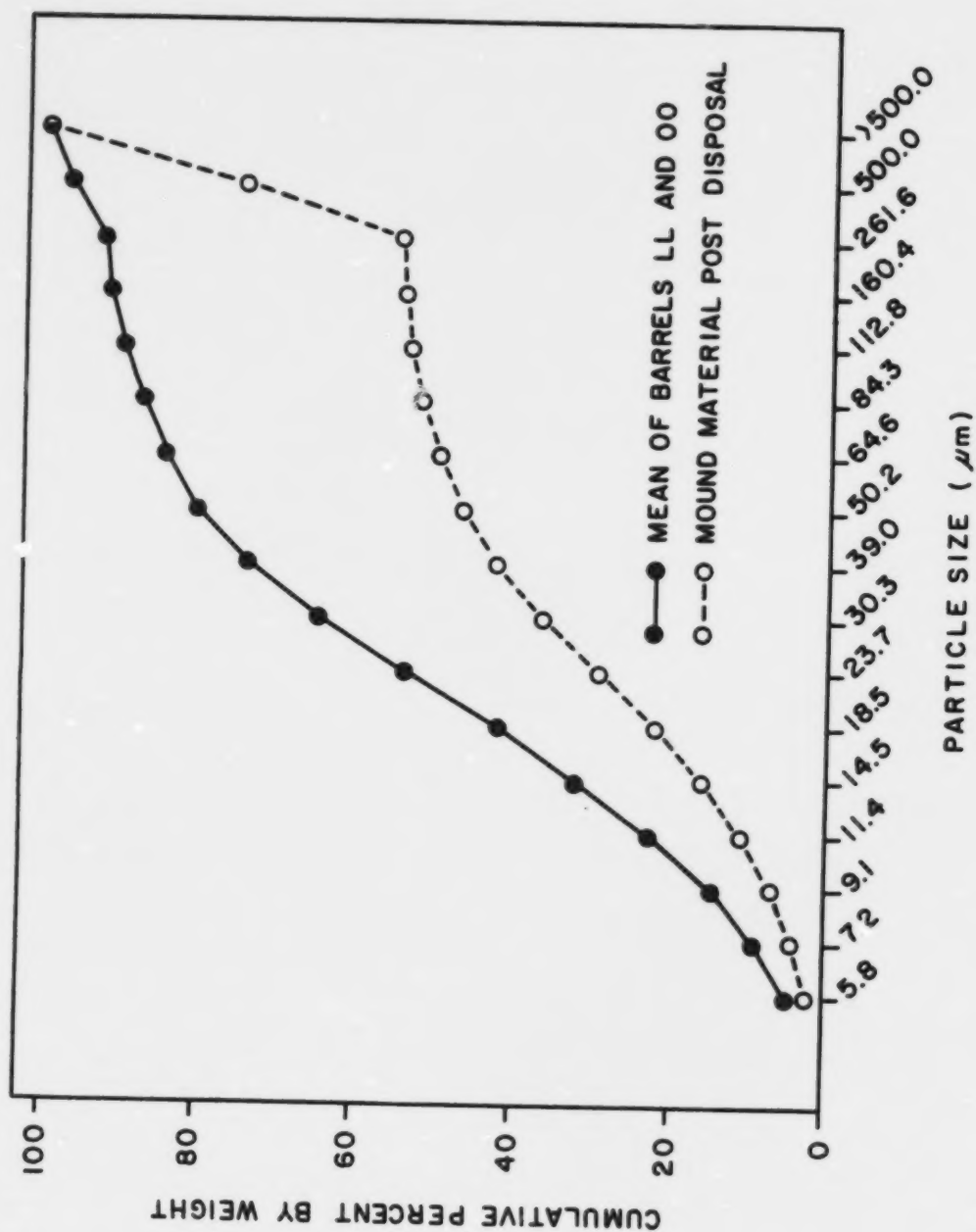


Figure 40. Particle size distributions of postdisposal CTR and BRH barrel sediments.

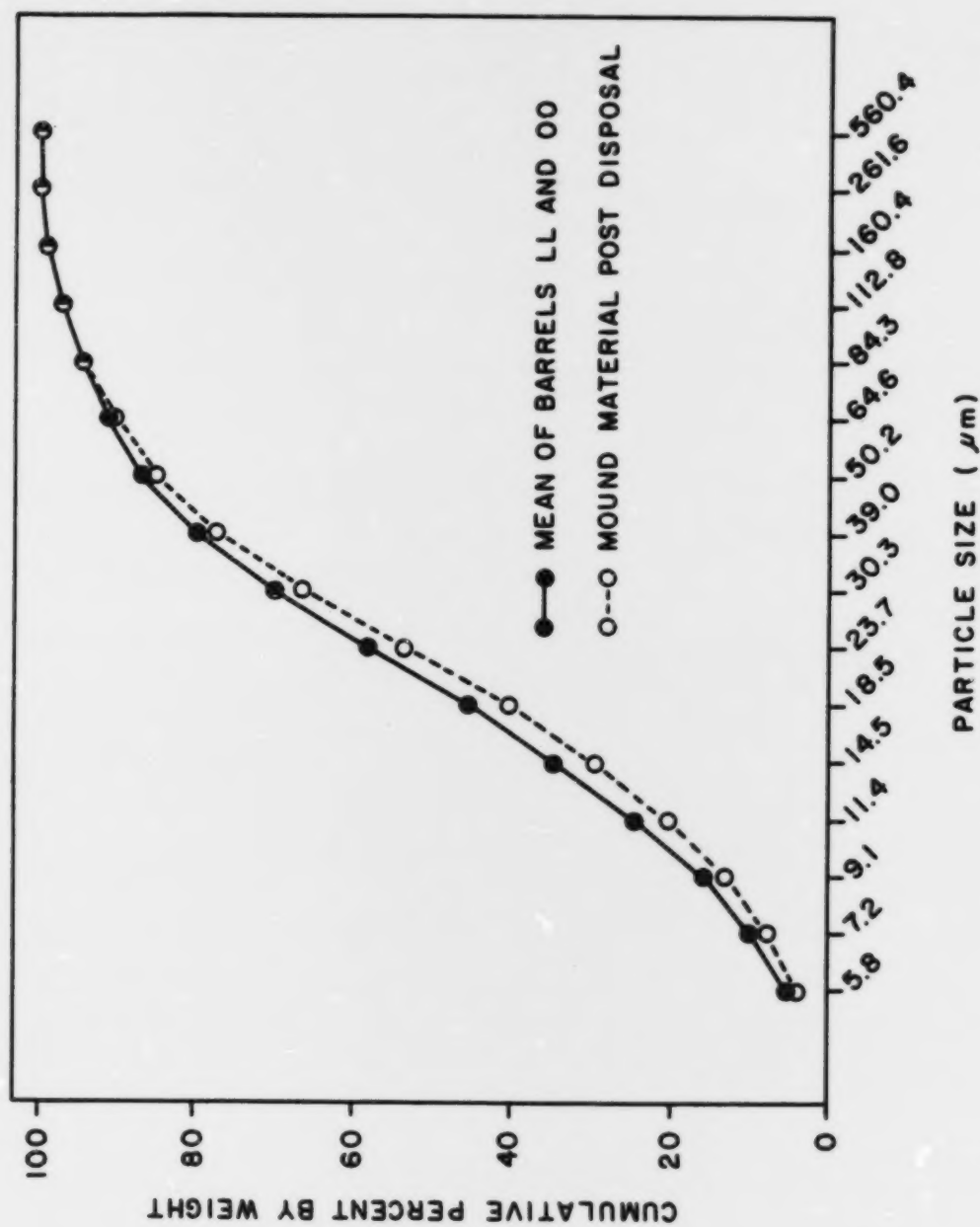


Figure 41. Particle size distributions of postdisposal CTR and BRH barrel sediments less than 560 μm .

Disposal of anoxic BRH dredged material resulted in a reduction in thickness of the oxidized layer in the vicinity of CTR. Figure 42 shows depths of the biological mixing zone (BMD) as measured by the REMOTS® benthic camera system in March 1983. BMDs during this predisposal period were between 3 and 5 cm at all stations. In June following the disposal operation, BMDs were depressed several cm on the mound relative to off-mound stations, as shown in Figure 43. Over the course of summer, BMDs on the mound increased at a rate of approximately 0.5-1.0 cm/month, although differences between CTR and other sites continued to exist through December 1984.

Representative images taken at CTR during REMOTS® surveys on 24 August 1982 and 13 June 1983 are presented in Figures 44 and 46, respectively. The most obvious difference seen between the sediments in these images was the presence of relatively dark, anoxic sediment in the latter. The highly reflective material in this image represents newly oxidized surficial sediment. The postdisposal image also lacks feeding voids which were apparent prior to disposal (Figure 44). A full presentation of the results obtained by benthic camera imagery may be found in Scott et al. (1984, 1987).

Summary

Atmospheric weather conditions during the study were fairly typical of those normally experienced in the LIS region. Air temperatures ranged between -18°C during December, January and February, and 35°C during July, August and September. No major storms occurred during the summer months following the disposal operation in May 1983; the first large storm occurred in mid-November 1983. Highest wind speeds in 1983 were recorded during a major storm event occurring in December, approaching 18 m/sec. Freshwater input to the Sound was seasonal, with 65% of the approximate 1.2×10^{13} ft³ of total annual surface

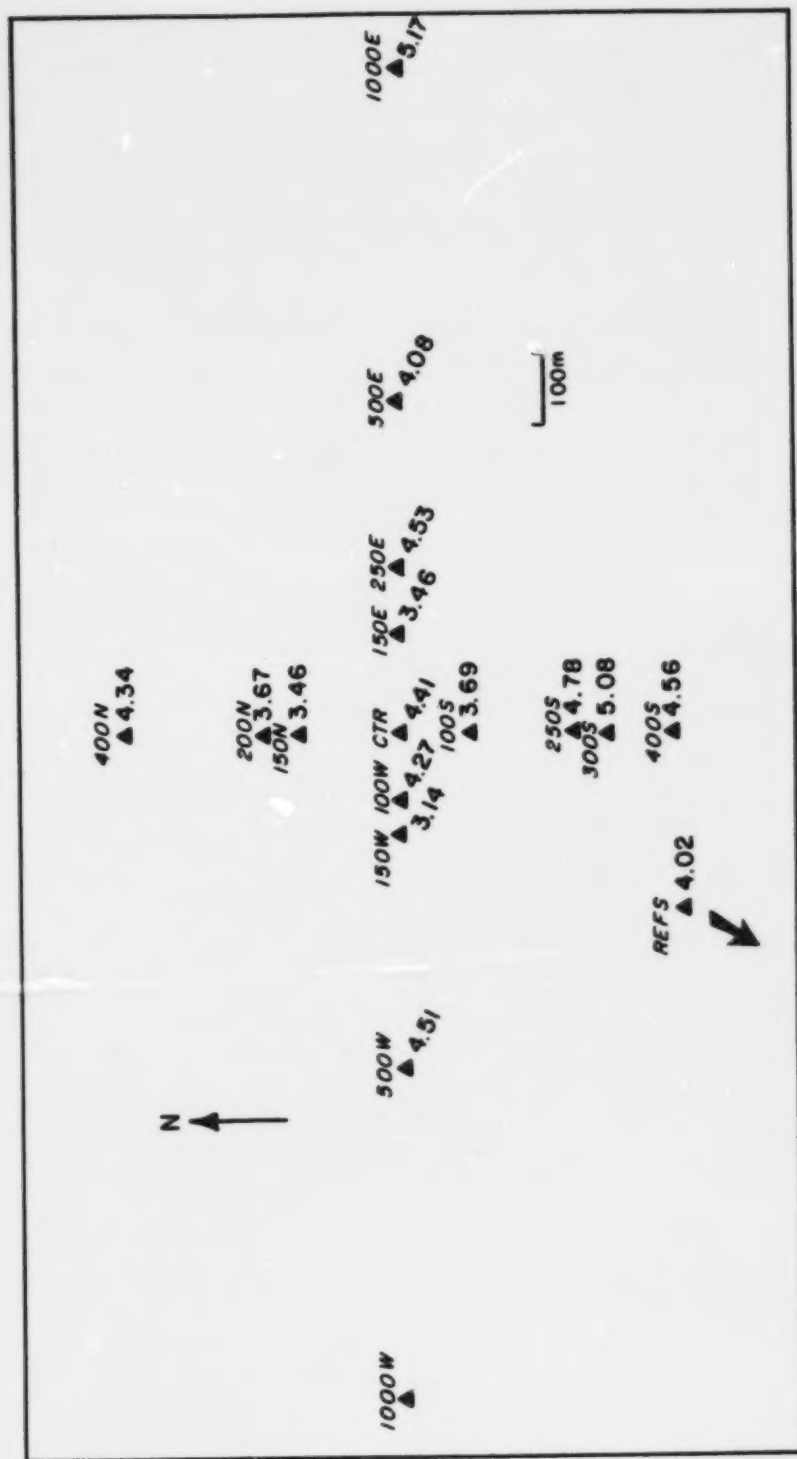


Figure 42. Depth of biological mixing in March 1983.

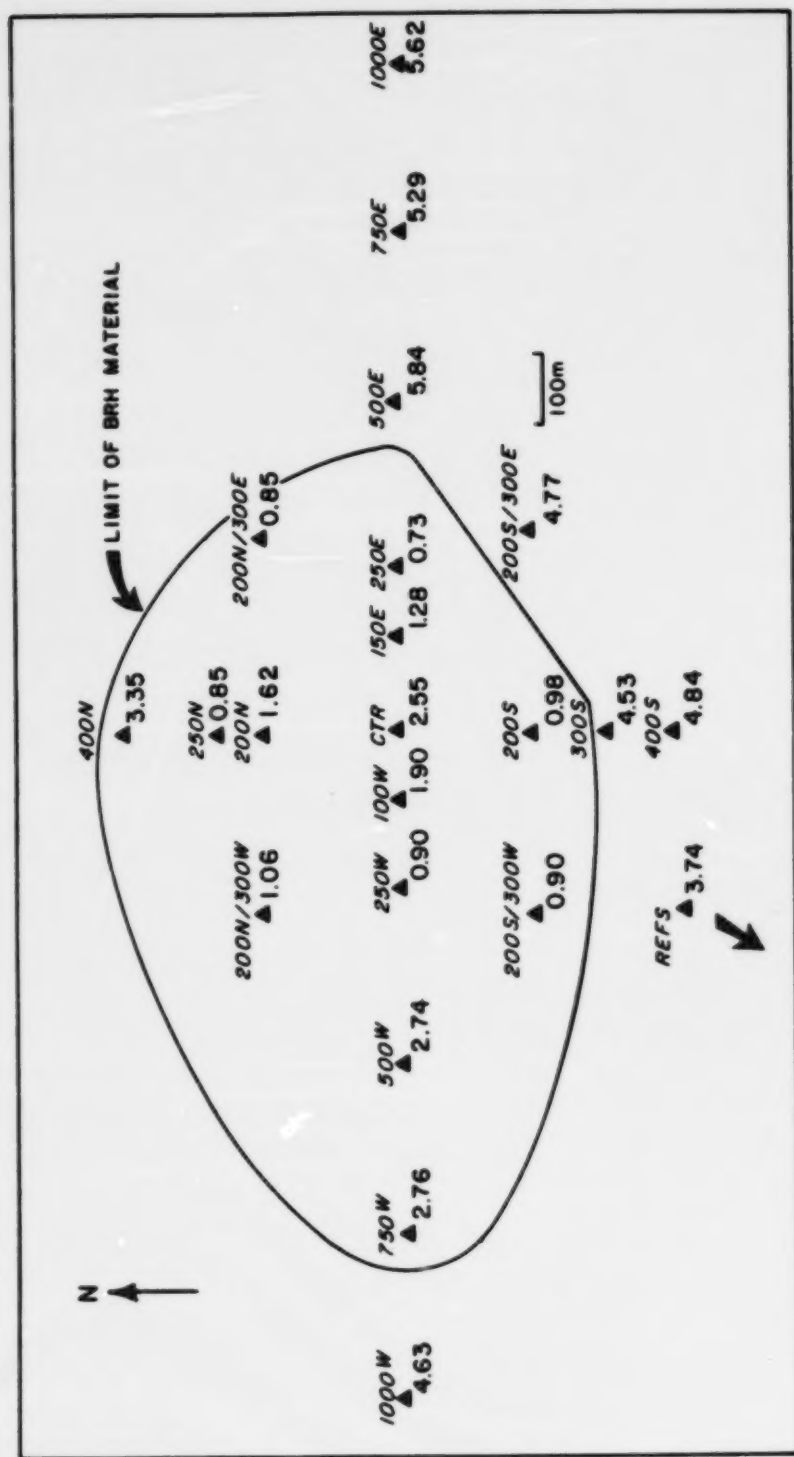


Figure 43. Depth of biological mixing in June 1983.



Figure 44. REMOTS image taken at CTR on 24 August 1982.



Figure 45. REMOTS image taken at CTR on 13 June 1983.

runoff occurring during the spring freshet from March through June. As no major drainage basins discharge water into the Sound from Long Island, NY, the majority of this runoff was derived from Connecticut river systems.

Tidal flow in the Sound is chiefly diurnal in nature, with tidal fluctuations averaging 2 m in the FVP study area. Overall circulation patterns are typical of estuaries: oceanic waters enter the Sound primarily at the eastern end through The Race on flood tide, are mixed with the less saline Sound waters, and leave through The Race on ebb tide. Tidally-induced currents are oriented primarily along the east-west axis of the Sound. Near-bottom current speeds in the FVP disposal area approached 0.6 knots along this axis, while maximum north-south speeds approached 0.4 knots. Based upon these values, a parcel of water could traverse several kilometers over the course of a single ebb or flood tide.

The FVP study area displayed a seasonal water temperature pattern, driven by air temperature and solar insolation. Thermal stratification occurred from April through October. Maximum surface and bottom temperatures were recorded in late summer, approaching 25 and 21°C, respectively. Winter temperatures were vertically uniform, approaching lows of 0-1°C. Surface and bottom salinities also displayed seasonal variation, produced in large part by freshwater runoff into the Sound. Surface salinities approached a minimum of 23 ppt during mid and late spring, and a maximum of 29 ppt during late summer and fall. Bottom salinities were less affected by freshwater input, reaching a minimum of 25 ppt and a maximum of 29 ppt. A pycnocline (at 7-10 m of depth) developed in spring and persisted throughout summer. Near-bottom dissolved oxygen concentrations ranged from greater than 10 mg/l in winter to 4-5 mg/l in late summer. Summer concentrations were 2-3 mg/l below saturation levels, apparently resulting from benthic respiration

and decomposition, and from isolation from surface waters by the seasonal pycnocline. The disposal operation and resulting disposal mound were not implicated in this DO depression, as levels at REFS were also low at this time of year. Small scale variability in all three parameters was observed to be produced by tidally-induced horizontal movement of inhomogeneous water masses through the study area.

A turbidity minimum was associated with the pycnocline during the summer months. Above this depth, turbidity was high as a result of primary production and surface runoff. Resuspension of bottom sediments resulted in high turbidity below the pycnocline. Suspended solids concentrations at 1 m above bottom were typically on the order of 5-25 mg/l. Variability in near-bottom suspended solids was observed through in situ instrumentation to be induced by tidal flow and aperiodic resuspension events, such as storms. Large excursions from background suspended solids levels, corresponding to the passage of turbidity plumes, were observed during the disposal operation. The activities of local commercial fishery operations also resulted in suspended solids peaks. Acoustic profilometer deployments revealed increasing turbidity with depth in the 1 m layer above the bottom, with levels near the sediment-water interface up to 30-50 times those measured at 1 m above the bottom.

Prior to the disposal operation, sediments around the FVP disposal area were relatively uniform, consisting chiefly of silt-clay sized particles, with a median grain size of 0.013-0.016 mm. Sediments collected from the FVP study reach were similar to these, although they displayed higher water content. Sediments collected prior to the dredging operation and stored in barrels for use in laboratory testing also displayed high water content, but were (apparently) more coarse than either the field sediments or the FVP study reach sediments. The

disposal operation produced a mound approximately 200 m in radius. This mound was initially 1.8 m thick at CTR, several cm thick at a distance of 100-150 m from CTR, and extended as a thin veneer to approximately 400 m. Thickness at CTR had decreased 20-40 cm one month following disposal, presumably resulting from tidal scouring and consolidation. Measured characteristics of postdisposal sediments were similar to those predisposal, with the exception of those at CTR and 200E. These two stations displayed decreased proportions of fine particles (with associated increased median grain size), along with decreased water content. CTR sediments continued to display high sand content through the spring of 1985. Vertical gravity cores taken in March 1985 at CTR indicated large variability between contiguous layers: relic clay clasts were interspersed with relatively clean sand and BRH material. The disposal of the anoxic BRH material resulted in a depression in the thickness of the oxidized layer in the vicinity of the disposal mound. This layer increased in depth at a rate of approximately 0.5-1.0 cm/month over the course of the summer following disposal, although differences between CTR and the other sites continued to exist through December 1984.

TRACE METAL CHEMISTRY

FVP Study Reach and Barrel Material

Trace metal concentrations in Black Rock Harbor sediment samples collected by NED prior to the dredging operation are given in Appendix E. These data cover the entire harbor region from the mouth (station BR-1) to the upper reach area (station BR-37). The FVP study reach was comprised of stations BR-8 through BR-32, and contained the highest concentrations of anthropogenic metals found in the harbor. There was, however, large variation in metal concentrations between stations in the study reach. Typically, concentrations of individual metals

differed by factors of up to 5 times over its length, with levels lowest near the mouth of the harbor. Mean concentrations of Zn, Ni, Pb, Cd, Cu and Cr are given in Figure 46.

Also shown in Figure 46 are the results of trace metals analyses of sediment samples taken from barrels during the rehomogenization process (see also Table 18). These data were used in part to determine the degree of variation present in the homogenized barrel material, and therefore the success of the rehomogenization process itself. Typically, this variation was low. Mn showed the largest coefficient of variation (CV, 12.5%), whereas Cd showed the smallest CV (2.4%). It can be seen in Figure 46 that the barrel material was fairly representative of the study reach and, barring major physical and chemical changes during the dredging, barging and disposal operations, representative of the material disposed at the FVP site in CLIS.

Sediment Chemistry

Prior to the disposal operation, trace metal concentrations in surficial sediments were fairly uniform throughout the FVP site, showing no significant differences between stations (ANOVA, $P > 0.05$). Mean concentrations at the biological effects stations are given in Table 19. These values compare favorably with metal concentrations reported for nearby stations by Greig *et al.* (1977; their station 69 at $41^{\circ} 9.0' N$, $72^{\circ} 50.0' W$) and Hunt (1979; his station 41 at approximately $41^{\circ} 9.0' N$, $73^{\circ} 5.0' W$), as shown in Figure 47. Also shown in this figure are the results of analyses performed in 1982 by NED. Based upon these studies, there have been no discernable trends in the concentration of trace metals over the past 6 years. Trace metal concentrations throughout the Sound obtained by Greig *et al.* (1977) are presented graphically in Appendix F for comparison purposes.

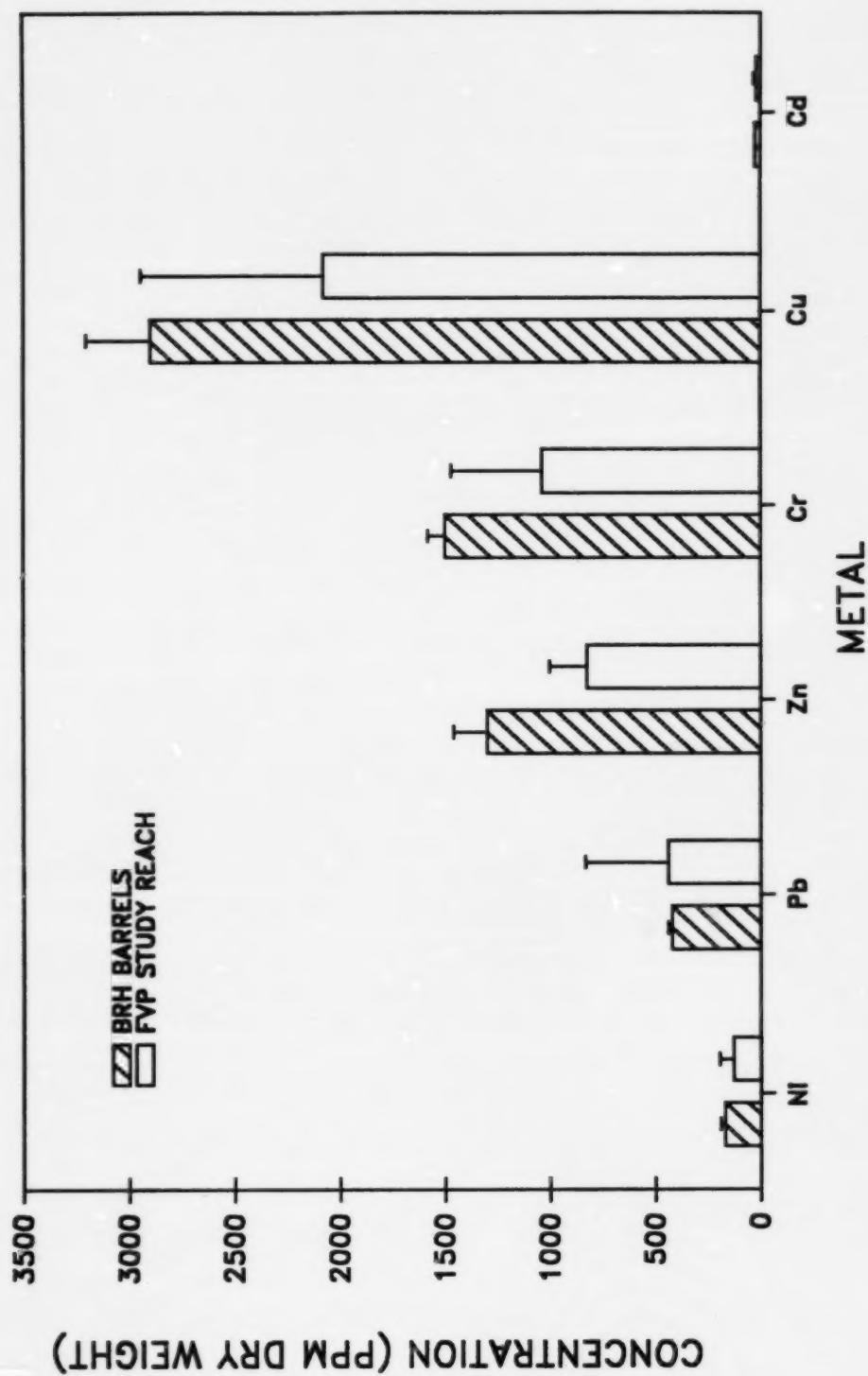


Figure 46. Trace metal concentrations in the FVP study reach and BRH barrel material.

TABLE 18. METAL CONCENTRATIONS IN BRH BARREL MATERIAL

Metal	Mean ($\mu\text{g/g}$ dry)	Standard error	Coefficient of variation (%)
Mn	310	16	12.5
Fe	30,800	1,140	8.9
Ni	170	8	9.2
Pb	420	8	5.4
Zn	1,300	65	12.4
Cr	1,500	33	5.6
Cu	2,900	122	10.9
Cd	24.7	0.2	2.4

TABLE 19. MEAN (SE) PREDISPOSAL TRACE METAL CONCENTRATIONS IN BIOLOGICAL EFFECTS STATION SEDIMENTS

Metal	CTR	200E	Station 400E	1000E	REFS
Mn	540 (79)	650 (146)	680 (163)	1,640 (40)	940 (28)
Fe	19,700 (680)	20,000 (1,450)	20,400 (930)	23,100 (140)	20,800 (590)
Ni	24 (<1)	24 (1)	25 (<1)	26 (<1)	24 (1)
Pb	51 (3)	52 (4)	54 (2)	57 (<1)	47 (<1)
Zn	140 (5)	140 (8)	150 (3)	160 (1)	140 (3)
Cr	51 (3)	51 (6)	53 (5)	59 (<1)	51 (2)
Cu	64 (2)	64 (4)	63 (1)	69 (1)	57 (1)
Cd	0.47 (0.01)	0.39 (0.05)	0.46 (0.23)	0.25 (0.02)	0.23 (0.01)

* Concentrations in $\mu\text{g/g}$ dry weight

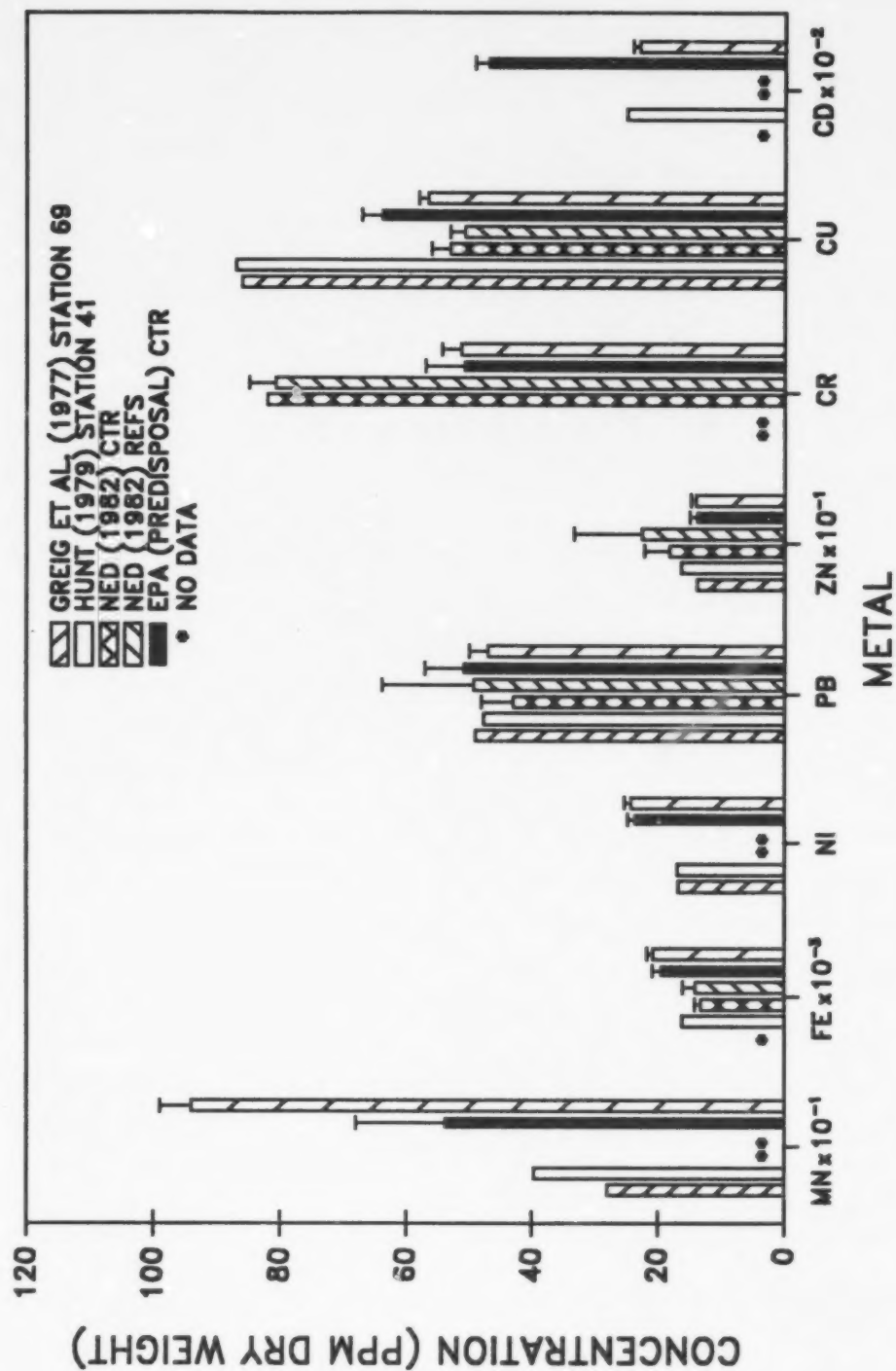


Figure 47. Trace metal concentrations in predisposal CLIS sediments.

Ratios of the metal concentrations measured in barreled BRH sediment to those in predisposal surficial sediments in the field are given in Table 20 and Figure 48. Concentrations of Cd, Cu, and Cr were greatly elevated (i.e. 62, 46, and 29 times, respectively) in BRH material relative to those in background CLIS, while Zn, Pb, and Ni were elevated to a lesser degree. Fe concentrations were approximately equivalent (1.5 times) between the dredged material and background CLIS sediments, and could therefore be used to normalize the concentrations of other metals to eliminate the influence of particle size (e.g. surface area, number of reactive sites) in comparisons of dry weight estimates of concentration. Mn was unique among measured trace metals, in that its levels were lower (0.4 times) in the BRH material. Thus, Cd, Cu, and Cr provided the strongest signals for tracking the distribution of dredged material in the sediments. In this discussion, descriptions of Cd will be presented as an example of the behavior of trace metals in the sediments. Information concerning other metals will be given when their behavior differs from that of Cd. All metal concentrations obtained from field sediments are given in Appendix G.

The influence of the disposal operation on trace metal concentrations at the FVP site can be seen in Figure 49. Open bars in this figure represent means of predisposal Cd concentrations at sampled stations, whereas shaded bars represent concentrations obtained in June 1983, immediately following the disposal operation. Although the mound of dredged material can be visualized by increased concentrations centered about CTR, concentrations at that station appear somewhat lower than those occurring at surrounding stations. Median particle size tended to be higher at CTR than at the surrounding stations (Figure 38). When individual concentrations are normalized to Fe (and therefore surface area and its effects), the expected peak in relative Cd concentration at CTR occurs (Figure 50). Thus, on

TABLE 20. TRACE METALS RATIOS OF BRH BARREL MATERIAL
VERSUS PREDISPOSAL SURFICIAL SEDIMENTS

Metal	BRH barrel*	CLIS predisposal*	Ratio
Mn	310	770	0.4
Fe	30,800	20,200	1.5
Ni	170	24	7.1
Pb	420	52	8.1
Zn	1,300	150	8.7
Cr	1,500	52	28.8
Cu	2,900	63	46.0
Cd	24.7	0.4	61.8

* concentrations reported as $\mu\text{g/g}$ dry weight

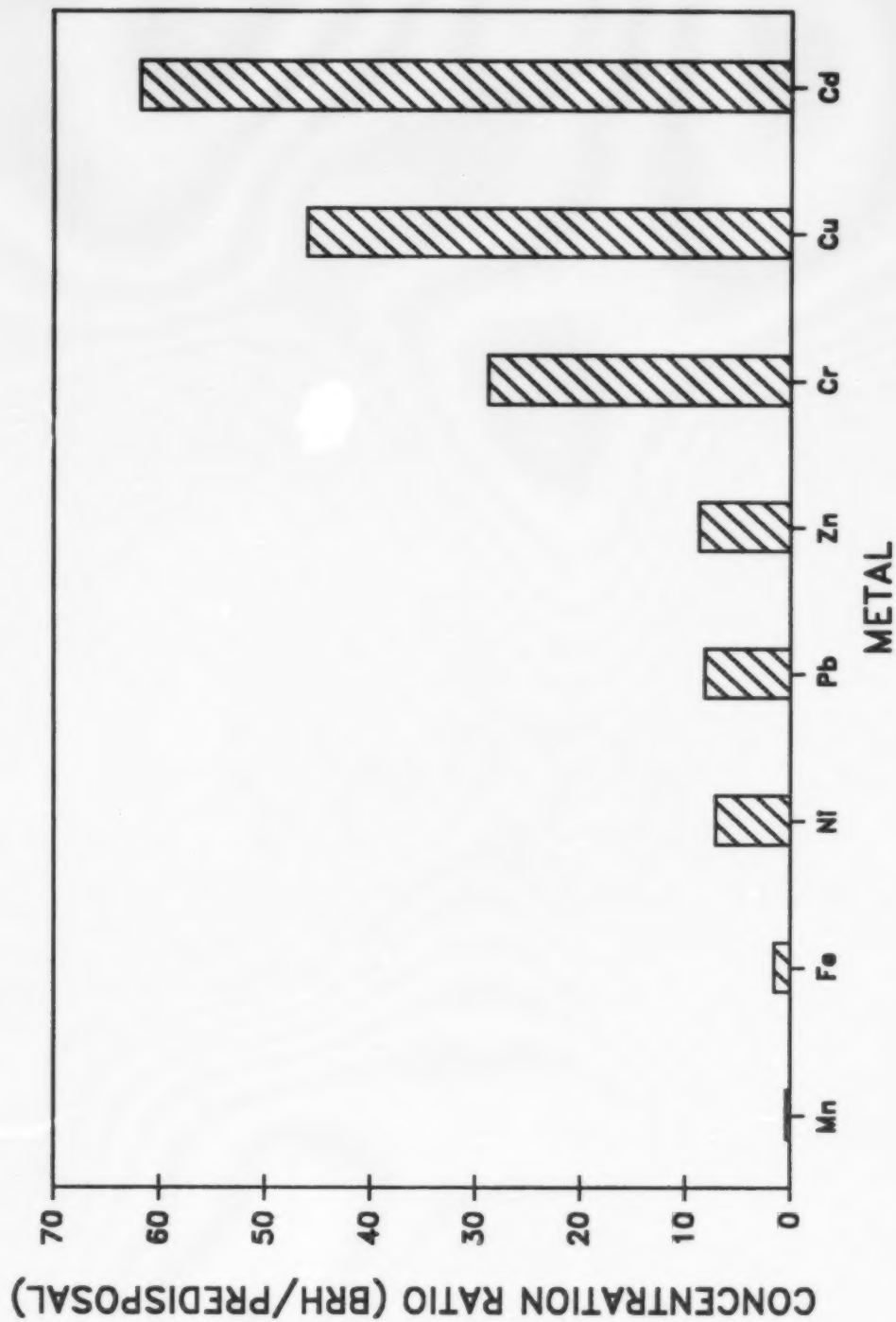


Figure 48. Ratios of trace metal concentrations in BRH barrel material versus predisposal sediments.

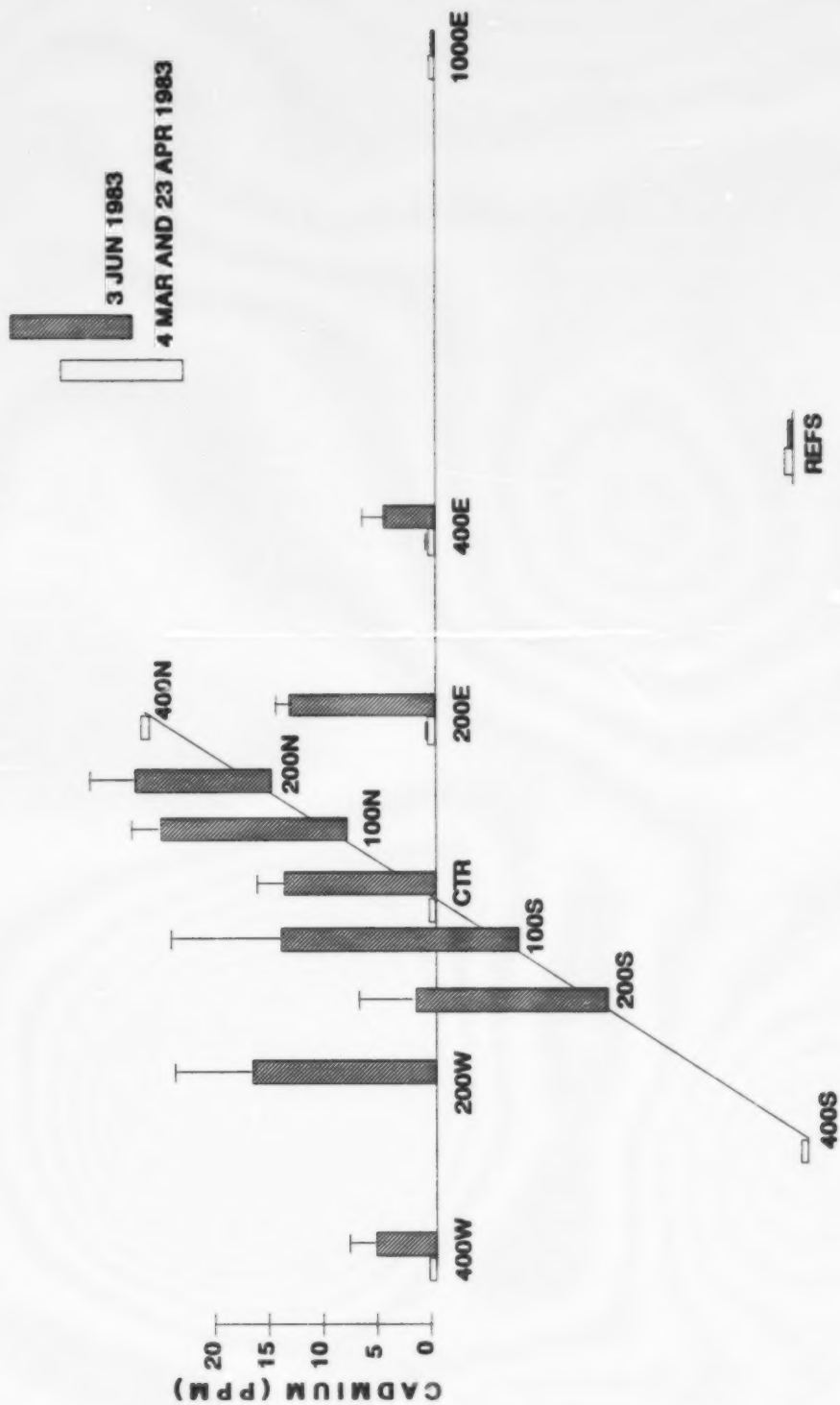


Figure 49. Cadmium concentrations in FVP field sediments before and after disposal.

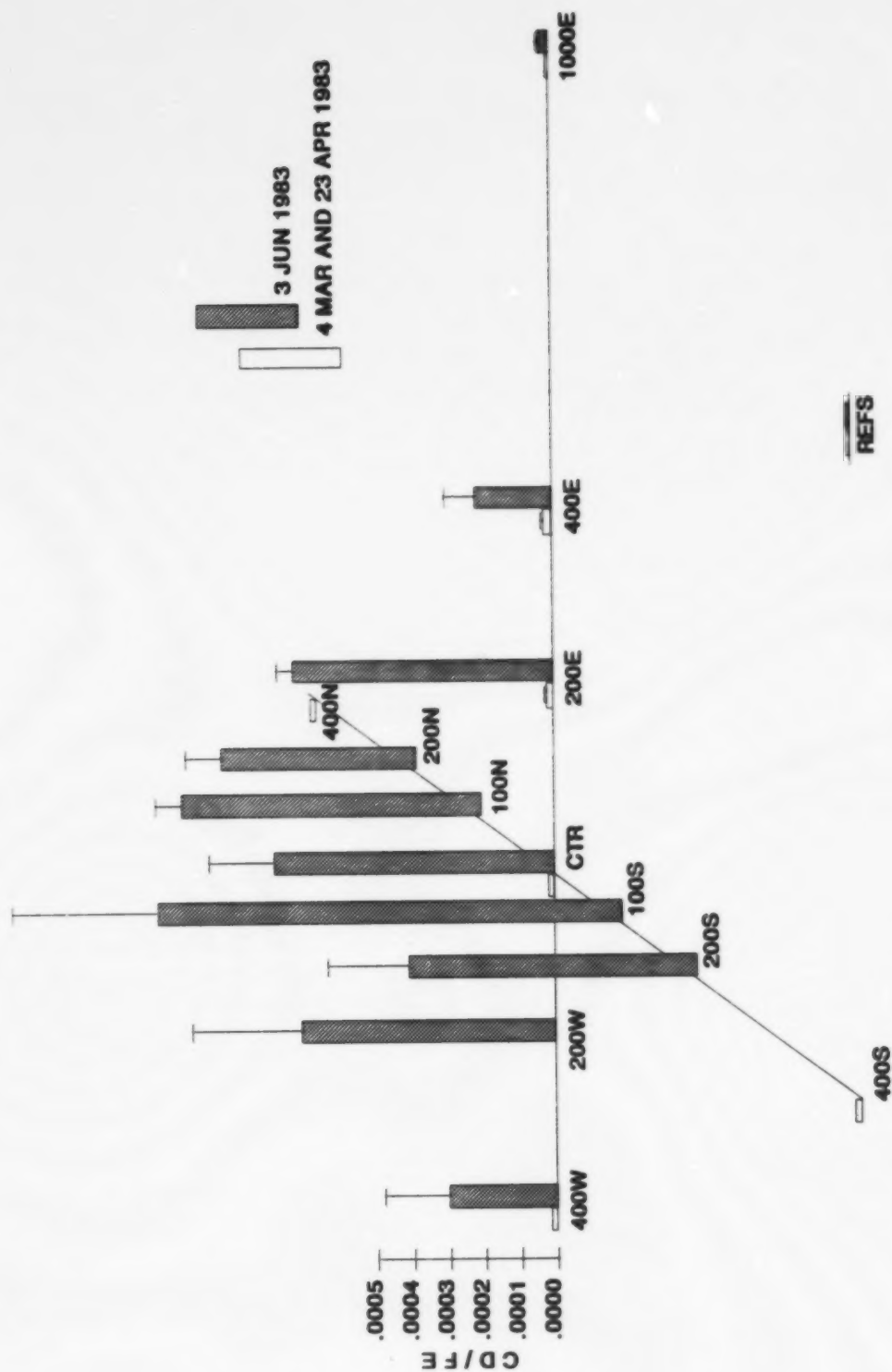


Figure 50. Ratios of cadmium to iron in FVP field sediments before and after disposal.

a normalized basis at least, the presumed gradient in trace metal concentration from CTR to 1000E appears to have existed.

With the exception of Mn and Fe, surficial concentrations of all metals within 400 m of CTR were elevated immediately following disposal relative to their predisposal levels (ANOVA, $P < 0.05$). As expected from the information presented in Figure 48, Mn levels were lower at mound and flank stations after disposal (ANOVA, $P < 0.05$), whereas Fe levels remained unchanged (ANOVA $P > 0.05$). Also, as expected, all metals (excluding Mn and Fe) displayed higher concentrations at mound and flank stations relative to REFS following the disposal operation (ANOVA, $P < 0.05$). Mound and flank Mn levels were slightly lower and Fe levels did not differ from those at REFS immediately following disposal. These data are given in Appendix G.

Despite the strong metal signals seen at mound stations immediately after disposal, surficial concentrations at those stations were only about 50% of those measured in BRH barrel material. Figure 51 gives calculated ratios of BRH metal concentrations to those in surface sediments at CTR, 200E, and 400E on 3 June 1983. Ratios for CTR and 200E all range between 1.2 and 2.7. Concentrations on the flank (400E) were even lower than would be predicted from the BRH material, so that levels of metals such as Cr and Cu were 6.5 times higher in BRH material. This apparent loss of contaminant mass is not due to underestimation of metal concentrations resulting from partitioning dynamics or postdisposal tidal scouring of fine particles as deeper horizons in the sediments of CTR and 200E were also lower than expected (see Appendix G). The larger ratios at 400E are probably a result of clean, underlying sediment being included in the scoop samples at that station. This information indicates the loss of up to a half of the contaminant mass during the dredging, barging, and disposal activities. Much of the loss may have resulted from retention of

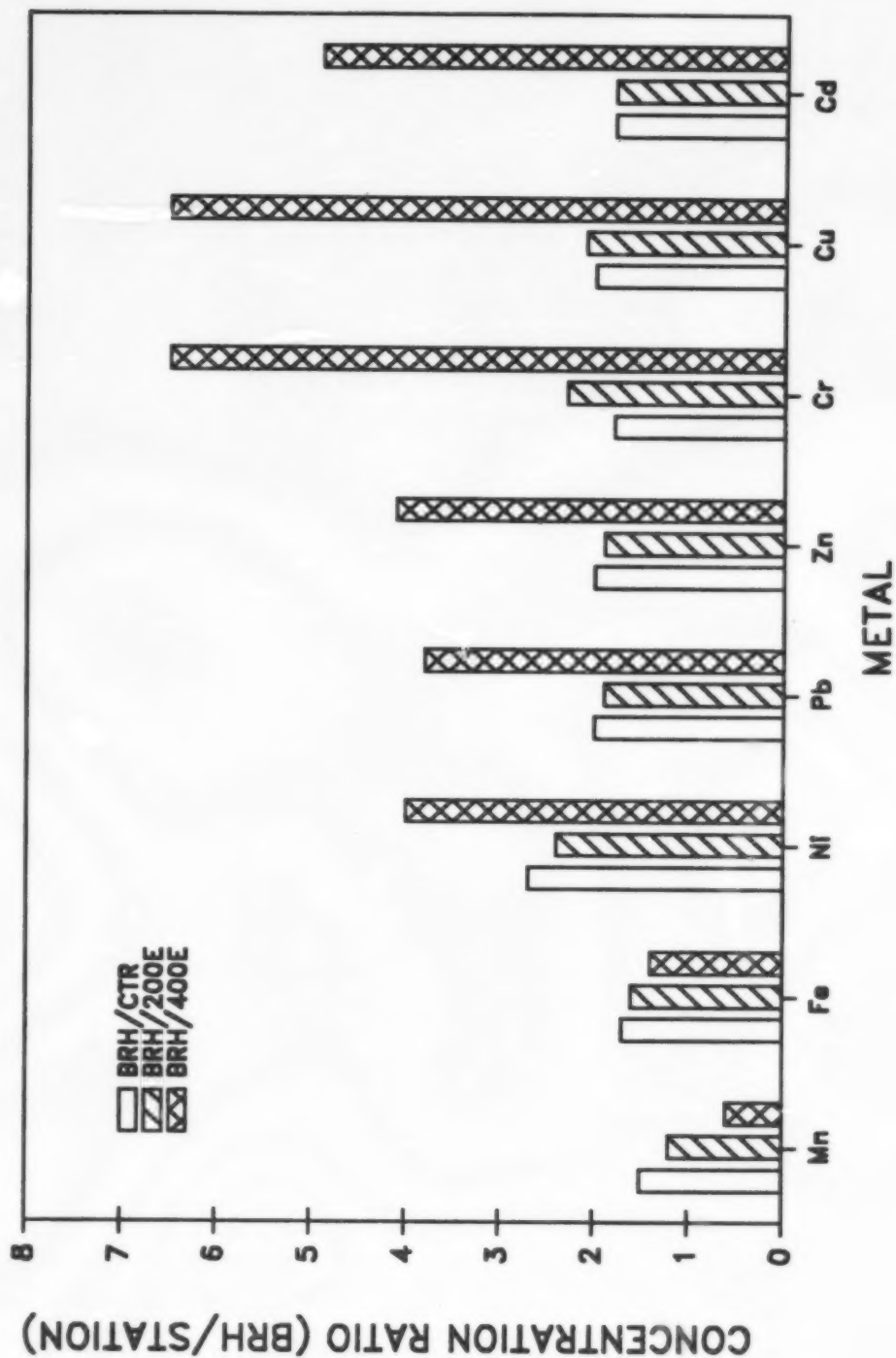


Figure 51. Ratios of trace metal concentrations in BRH barrel material versus postdisposal sediments.

fine particles (with relatively high surface area to volume ratios) in the water column during dredging and disposal.

Trace metal characteristics of REFS typically remained unchanged over the course of the study, as can be seen in Figure 52 for Cd. Although variation on the order of 10 to 20% was evident, no trends were observed over time or with depth (recall that sediment cores collection began in July 1983). In contrast, surficial Mn concentrations decreased by approximately 40% immediately following disposal (Figure 53), and remained low over the course of the study. Although this decrease is suggestive of BRH contamination at REFS (Mn levels in the dredged material were 0.4 times those in background CLIS sediments (see Figure 48)), we do not know whether the apparent decrease was due to sampling error, or if in fact Mn displayed some dynamic associated with either the disposal operation or mobilization differences associated with changes in redox conditions and summer warming (e.g. Calvert and Price, 1972; Hem, 1963). Similar changes in surficial Mn levels were observed at all stations sampled (see Appendix G). The generally higher surficial concentrations of Mn relative to those deeper in the sediment may result from the solubility differences of this metal under oxidized versus reduced environments (W. Boothman, SAIC, personal communication).

The elevation in metal concentrations at the mound and flank stations (Figures 49 and 50) continued throughout the two year period following disposal, as can be seen for surficial Cd in Figure 54. There was, however, a general reduction in surficial metal levels during this period. Mobilization of particulate-bound contaminants via oxidation reactions and partitioning dynamics following disposal may account for much of this reduction. Additionally, dilution of the disposed material by deposition of clean, background sediments may also have occurred. However, Cd levels remained noticeably elevated as late

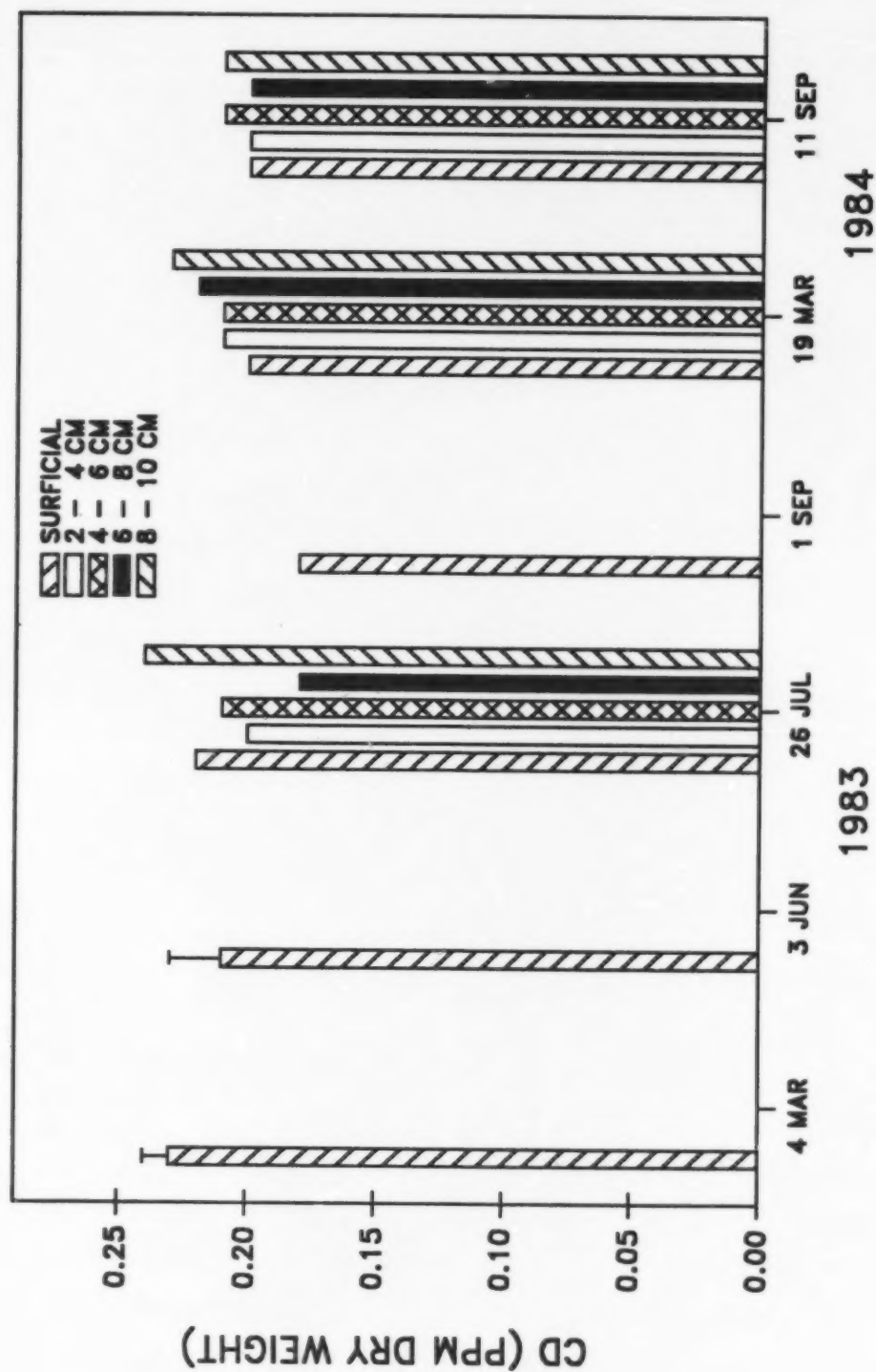


Figure 52. Depth profiles of cadmium concentrations in REFS sediments over time.

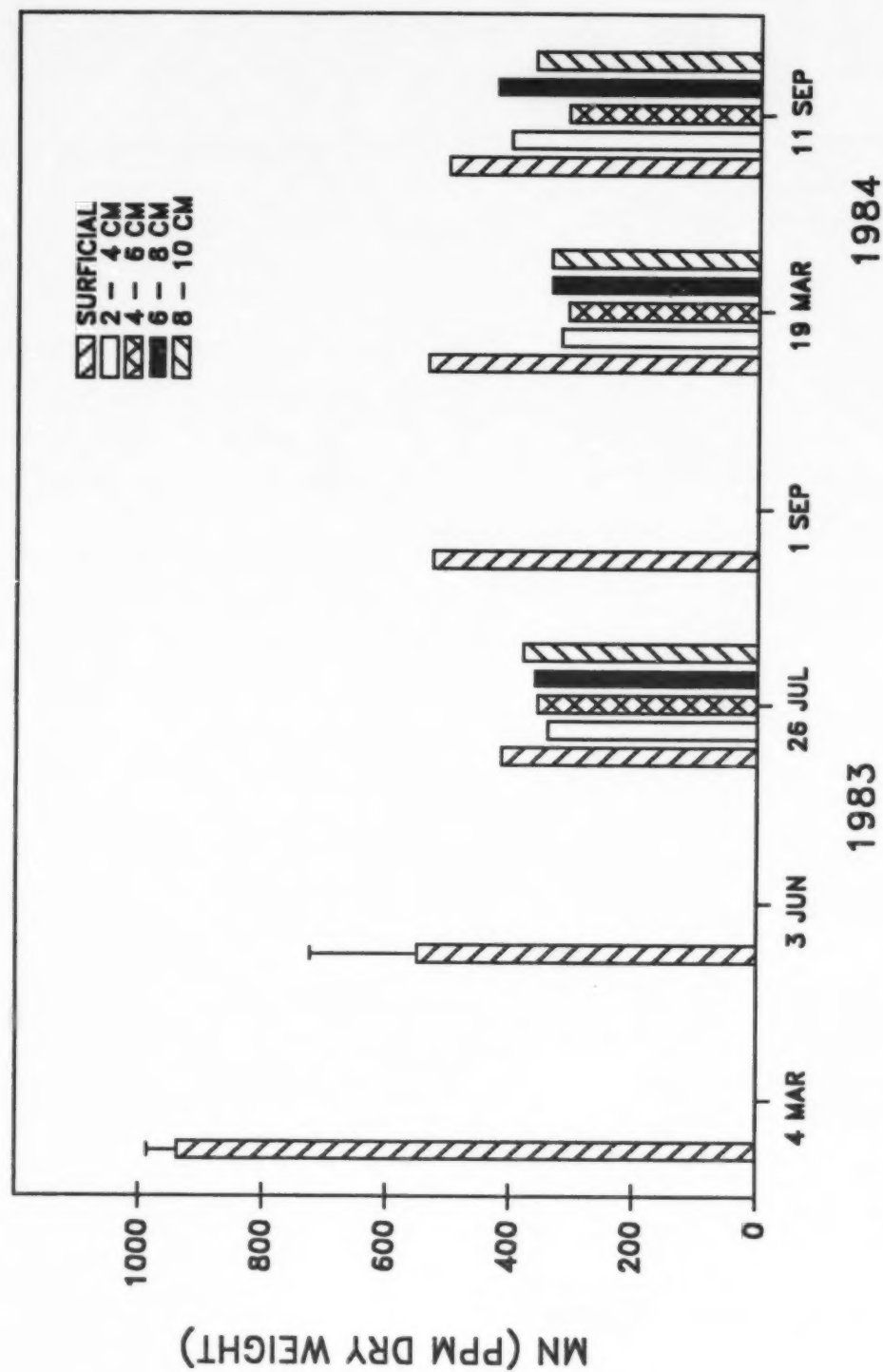


Figure 53. Depth profiles of manganese concentrations in REFS sediments over time.

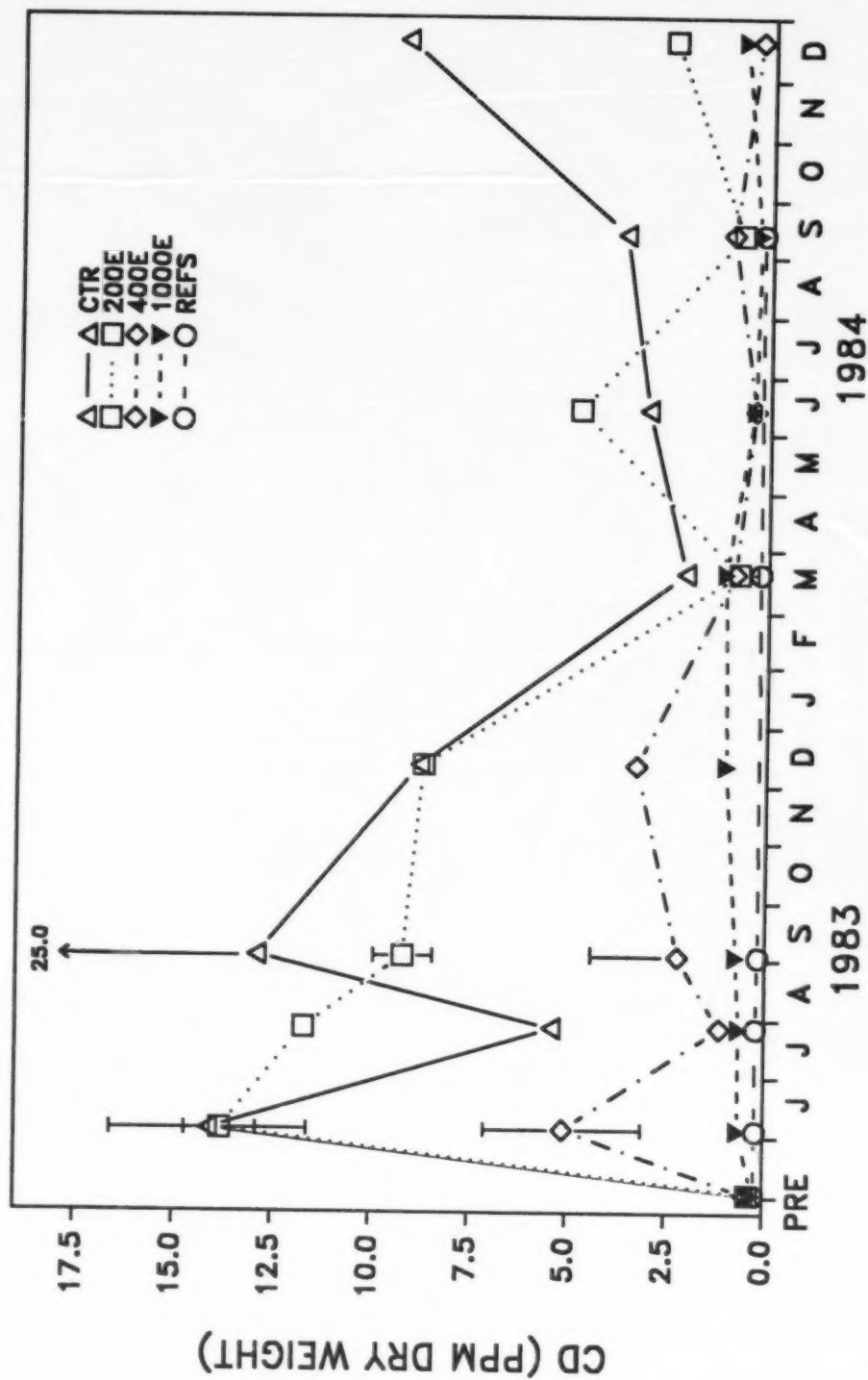


Figure 54. Cadmium concentrations in biological effects station surficial sediments over time.

as December 1984.

Figure 54 indicates a high degree of variability between dates at the mound and flank stations. Much of the variation may be accounted for by inadequate sampling of spatial variation in contaminant distributions. An analysis of Cd levels at various depths within the gravity cores collected from CTR in March 1985 indicated differences of 2-3 orders of magnitude occurring within 5 cm. Presumably, this degree of variation was also present horizontally. Extreme heterogeneity may have resulted from variation in metal concentration between barge loads of dredged material and the degree of mixing of material on the sea floor.

Increased Cd concentrations following disposal were also observed at 1000E (Figure 54). This elevation was restricted primarily to the upper layer of sediment, as indicated in Figure 55. Other metal species displayed similar increases as well. This unexpected contamination of 1000E sediments may have resulted from initial transport of dredged material during the disposal operation, from secondary off-mound transport of resuspended particulates, or from an errant dump of dredged material occurring nearby. Support for this final possibility comes from the observation of apparent dredged material at a site approximately 50 m to the northwest by Feng and DeGoursey (1984) during a related study. The persistently high (and apparently increasing) Cd levels through March 1984 suggest continued transport of dredged material to the site from either the FVP mound or the site of the presumed errant dump. Based upon pre- and postdisposal Cd concentrations at 1000E and BRH barrel levels, 1.5-3.0% of the upper 2 cm of sediment at that station could have been dredged material. The apparent cycling of Cd levels following the summer of 1983 (i.e. high in winter, low in summer) may have been the result of continued off-mound transport combined with changes in the activity of benthic fauna

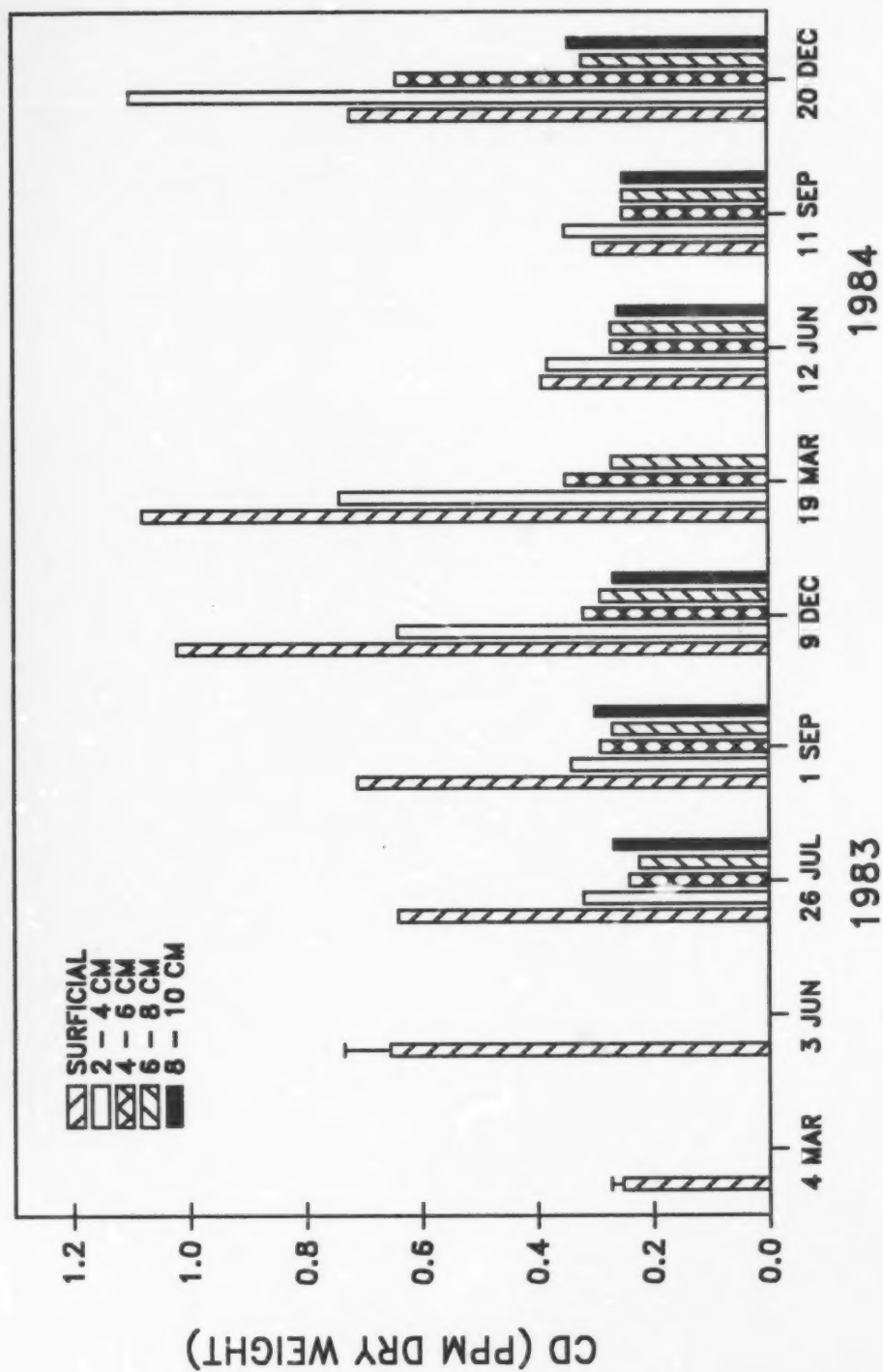


Figure 55. Depth profiles of cadmium concentrations in 1000E sediments over time.

(e.g. Rhoads, 1963, 1967, 1970; Rhoads and Boyer, 1982; Davis et al., in prep.).

Water Column Chemistry

Trends in time and space--

Trace metal dynamics in the water column are well represented by Cd and Pb concentrations, as Cd is relatively soluble under ambient CLIS conditions, whereas Pb is relatively insoluble. As both elements displayed increased concentrations in the dredged material relative to background CLIS sediments, influences of the disposal operation should be detectable through their examination. The following discussion of inorganic water column chemistry will therefore focus on these two metals. Complete listings of dissolved, particulate and whole water trace metal concentrations are given in Appendix G.

Predisposal levels of Cd measured in water samples collected over a 10 hour period at CTR are shown in Figure 56. Both near-bottom and 5 MBS concentrations are initially high and decrease with time. Near-bottom concentrations at REFS, the other station sampled on that date, were 0.11 ± 0.001 (1 SE), approximately the level reached at the end of the sampling period in Figure 56. This variation appears to be related to the movement of heterogeneous water masses through the FVP study area. Low tide on this date (22 April 1983) occurred at approximately 13:00, with the succeeding high tide occurring at approximately 19:30. Thus Cd levels decreased as resident water was replaced by that previously residing to the east of CTR, outside of the CLIS disposal area. This suggests the existence of a 'cloud' or plume of suspended dredged material produced by disposal operations ongoing at other sites within CLIS. The pattern of near-bottom Cd change is not due simply to changing suspended solids concentrations, as the pattern remains when concentrations are reported per dry weight of suspended solids

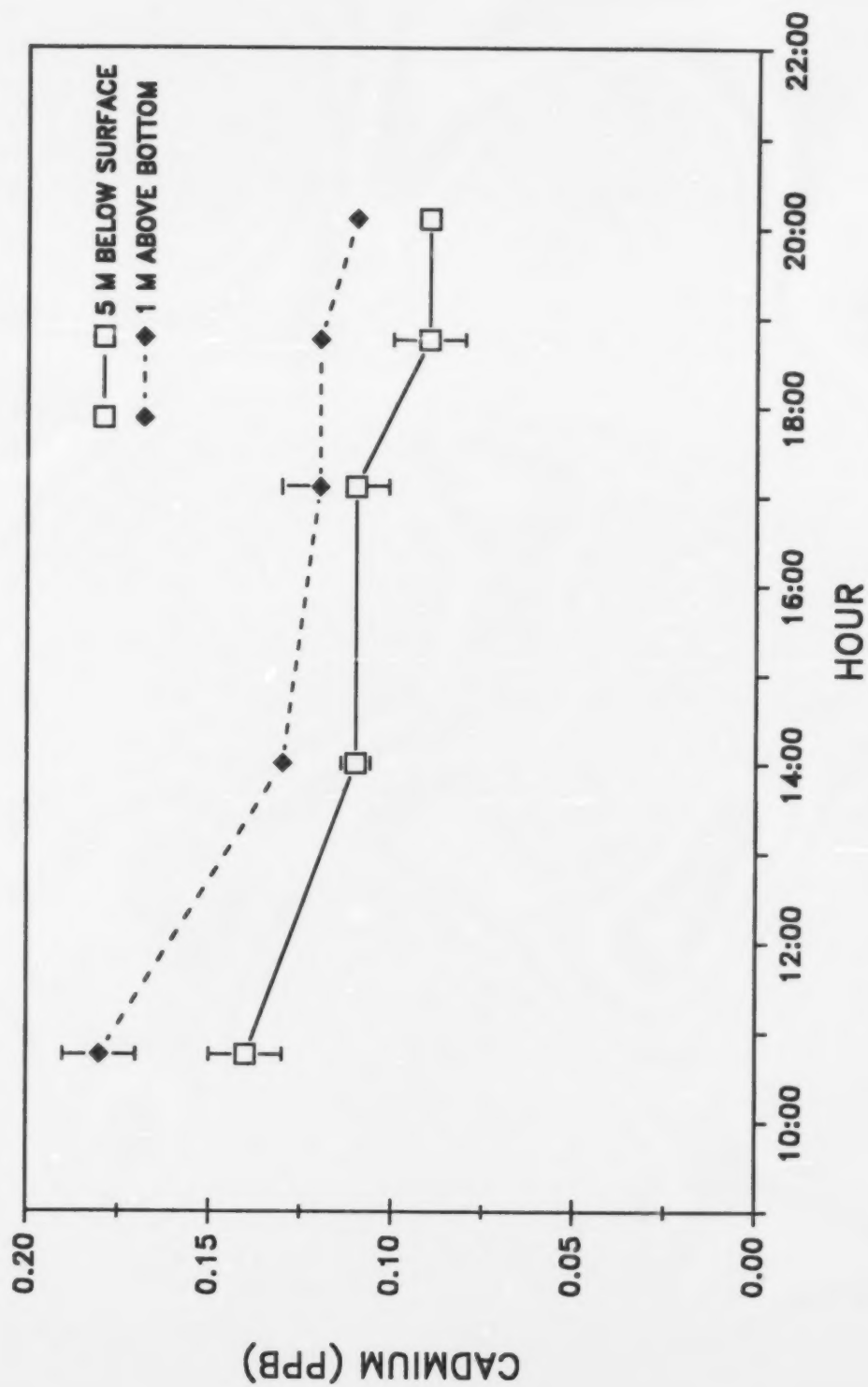


Figure 56. Cadmium concentrations in predisposal CTR whole water samples.

(Figure 57). With the exception of Mn, all other metals showed similar trends. Mn displayed the opposite pattern, again supporting the idea of a plume of dredged material with characteristics similar to BRH material.

Near-bottom whole water concentrations of metals measured at CTR on 7 June 1983 (one month following cessation of the disposal operation) were generally greater than those at REFS (Duncan's multiple range test, $P < 0.05$). The only exception in this trend was Ni, which did not differ between the two stations. Figure 58 shows levels of Cd and Pb for the biological effects stations at which samples were collected. Although detectable, these signals were generally weak, such that the values at CTR were less than twice those at REFS (Figure 58). Elevated levels (relative to REFS) could also be seen at both 400E and 1000E in Cd, Zn, Cu, and somewhat surprisingly, Mn. With the exception of Mn, these differences remain when concentrations are corrected for suspended solids concentration (see Figure 59), indicating the presence of dredged material in the water column. Corrected levels of Mn were lowest at CTR ($P < 0.01$). No differences were found between stations in solids-corrected surface waters ($P > 0.05$).

Because the signal of trace metals in the water column was very weak and affected by the concentration of suspended solids, evidence of dredged material in the water is best detected as increased concentration beyond that due to particulate concentration. Whole water Cd and Pb are regressed against suspended solid concentration in Figures 60 and 61. All samples are included in these plots. Although the regression relationship for Cd is significant ($P < 0.05$), the coefficient of determination was extremely low ($r^2 = 0.05$), as is expected for a chemical displaying little affinity for the particulate phase. The upper and lower curves in these figures represent the 95% confidence interval about the regression line. Three points are

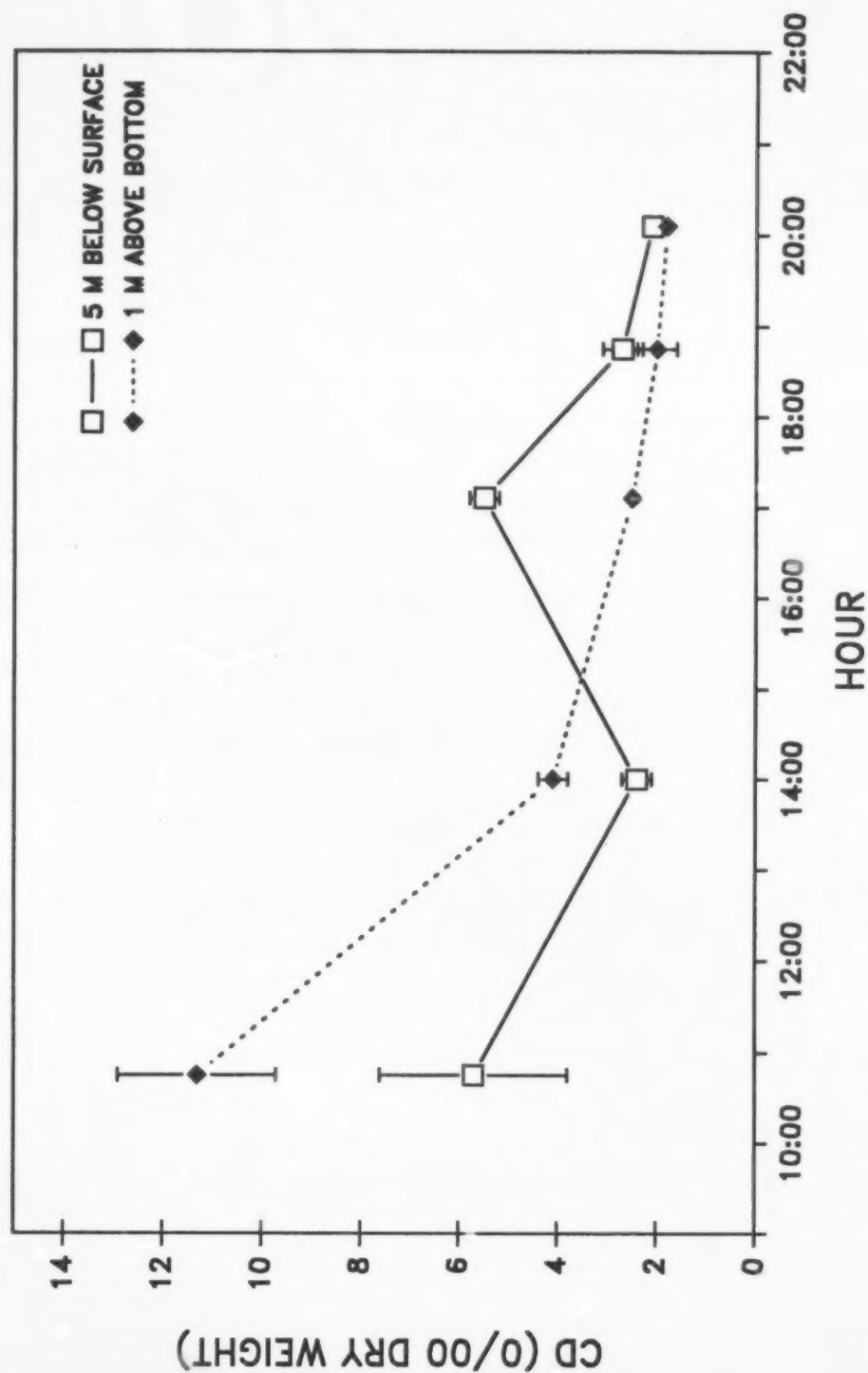


Figure 57. Particulate-associated cadmium concentrations in predisposal CTR water samples.

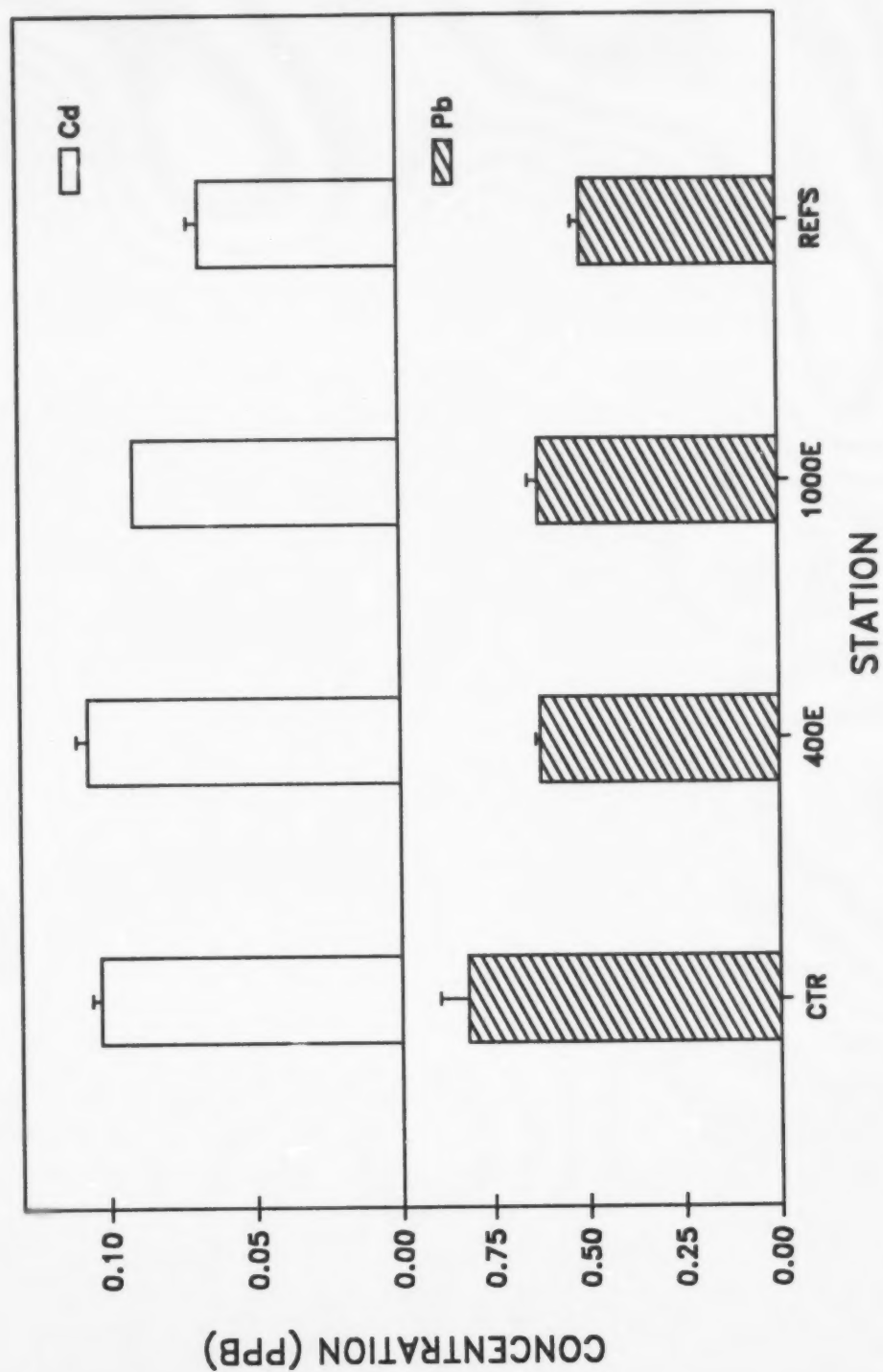


Figure 58. Cadmium and lead concentrations in postdisposal whole water samples at biological effects stations.

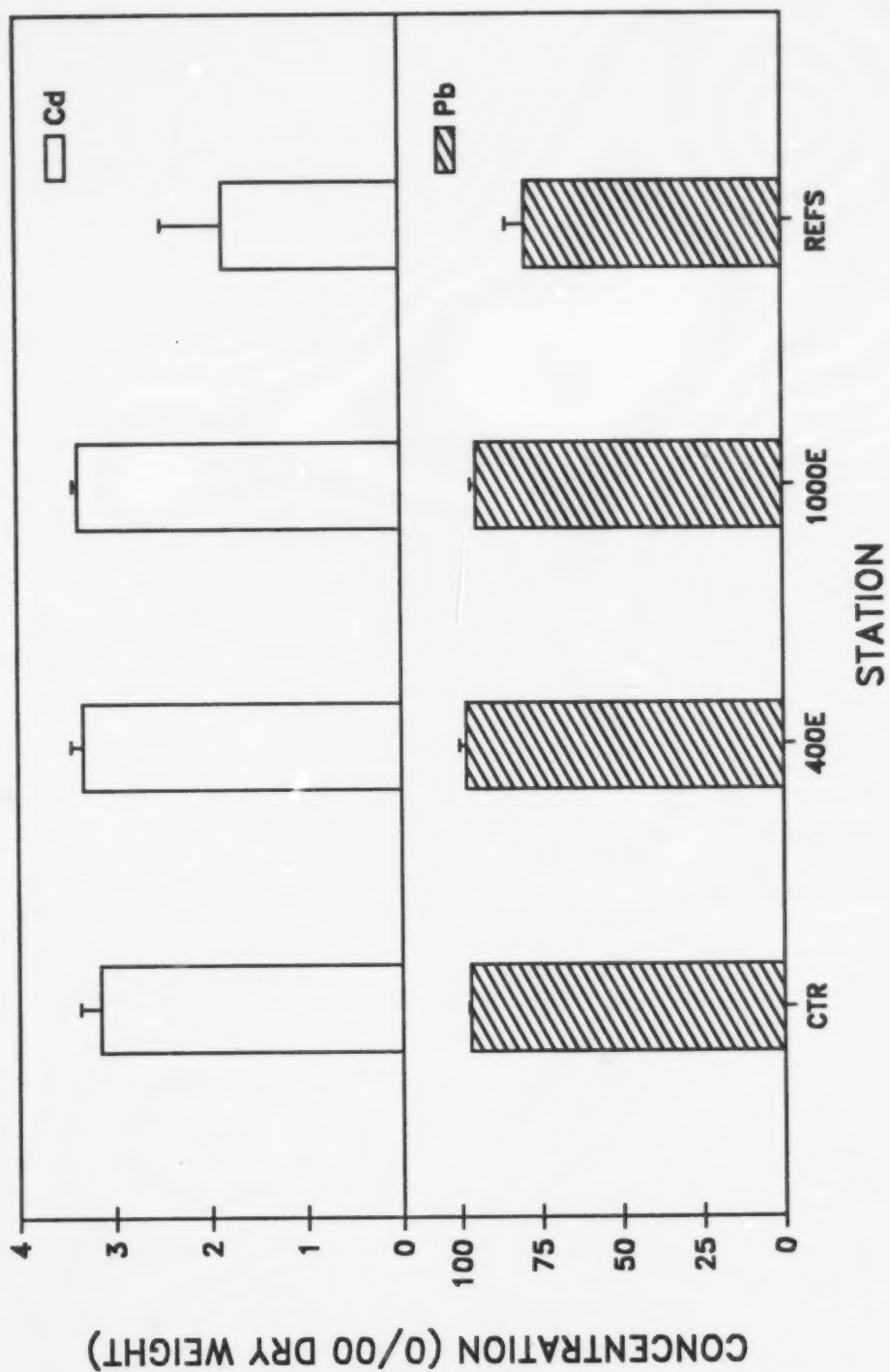


Figure 59. Particulate-associated cadmium and lead concentrations in postdisposal water samples at biological effects stations.

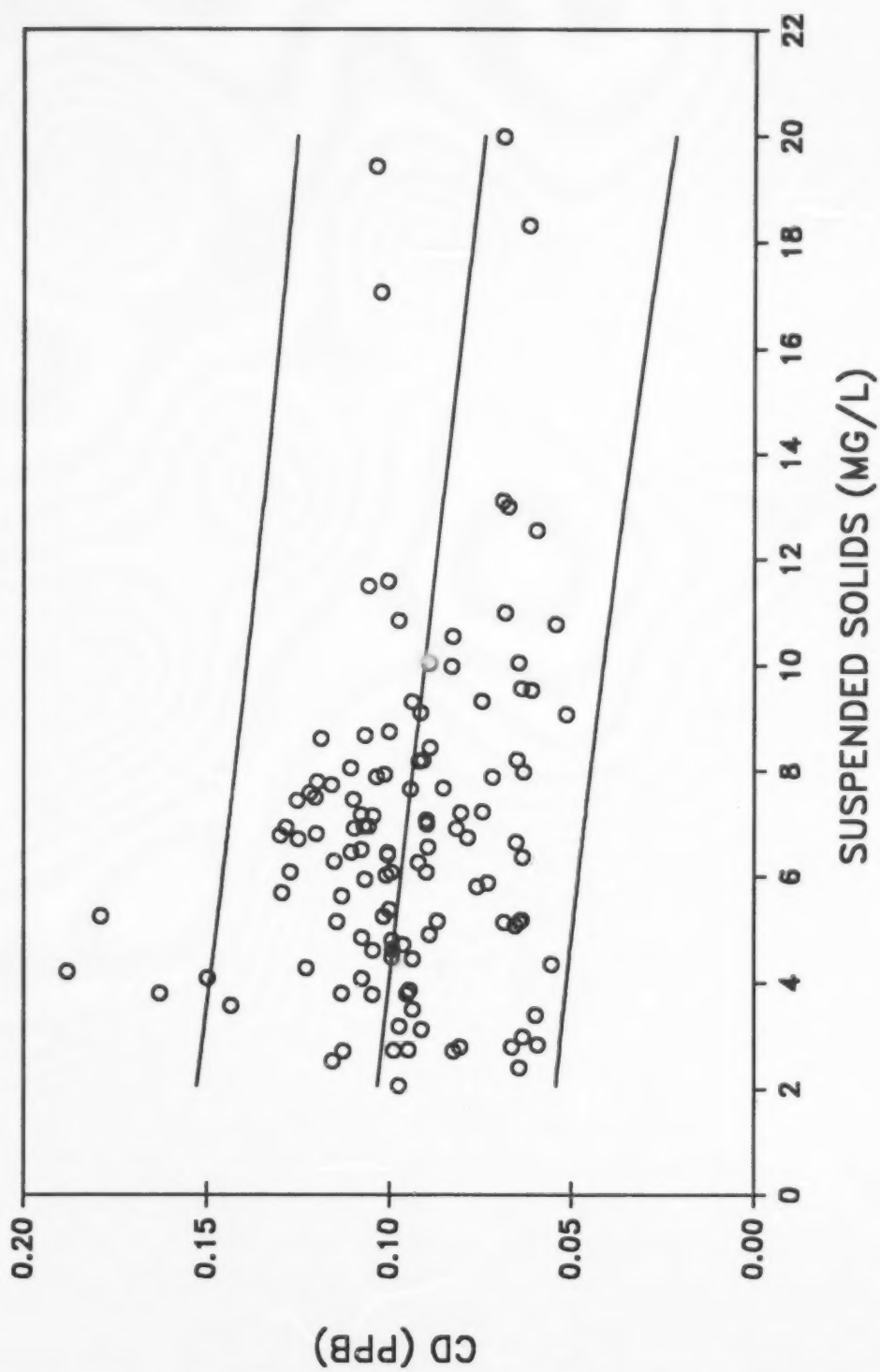


Figure 60. Whole water cadmium versus suspended solids concentration.

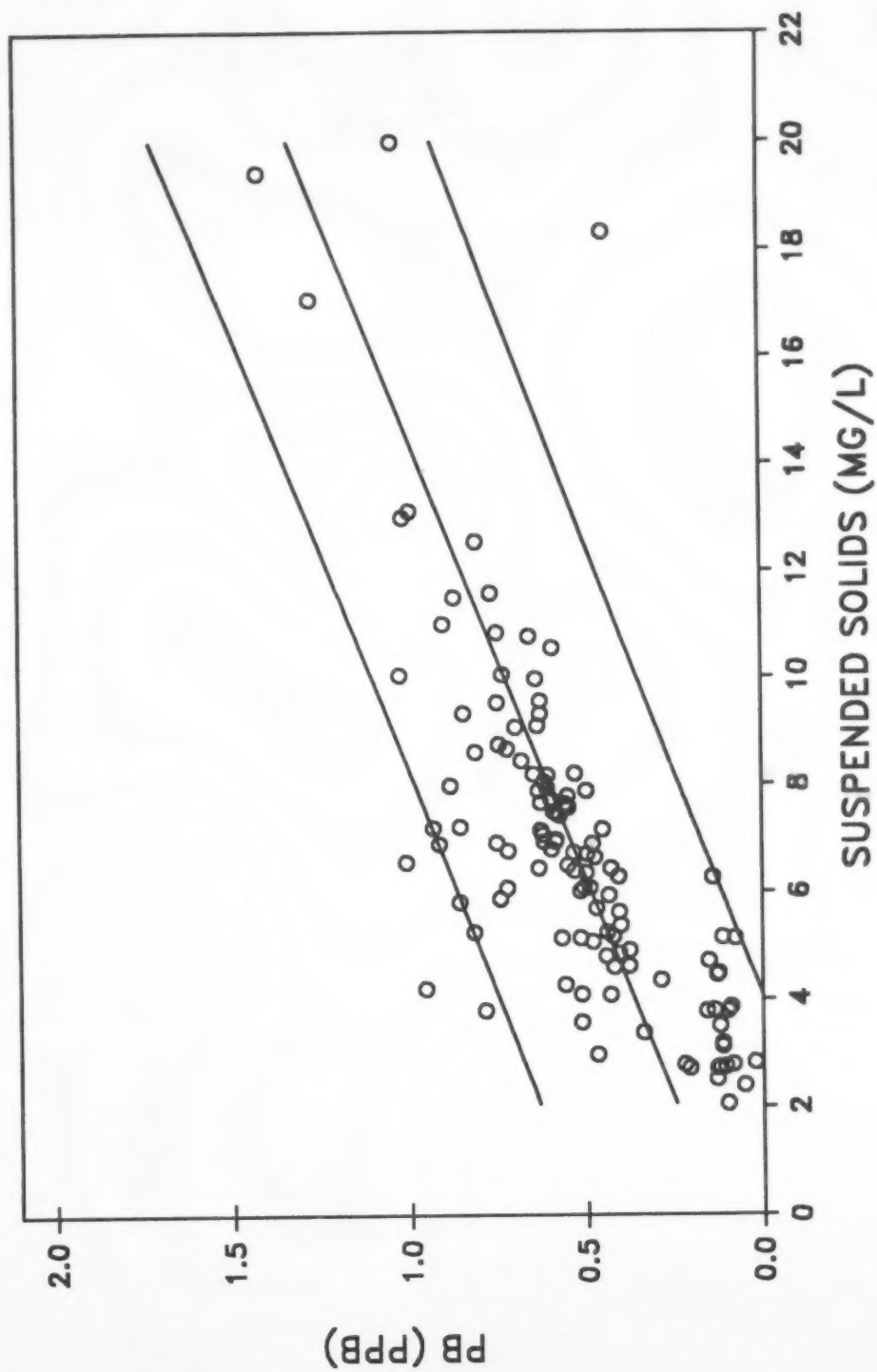


Figure 61. Whole water lead versus suspended solids concentration.

obvious outliers in Figure 60. These represent Cd levels measured near-bottom at 10:45 on 22 April 1983 at CTR. The mean of these replicates is also shown in Figure 56. Pb's relatively strong affinity for the particulate phase is demonstrated in Figure 61. The regression is highly significant ($P < 0.0001$), and explains 53% of the variation in Pb concentration. Four suspected outliers, three with elevated Pb levels and one with depressed levels, are evident in this figure. The three elevated concentrations were measured near-bottom at 10:45 at CTR (two replicates) prior to disposal, and near-bottom at 400E on 6 June 1984. The depressed concentration occurred near-bottom at 200W on 6 June 1984. Although we have no explanation for the low 200W sample, the single elevated level measured one year subsequent to disposal at 400E argues against large-scale transport of trace metals off the disposal mound once it had formed. However, somewhat elevated Cu levels were observed in two CTR replicates collected near-bottom on 7 June 1983.

Phase partitioning--

Partition coefficients (K_p) were calculated for the four metals (Cd, Cu, Ni, and Mn) for which both dissolved and particulate measurements were made. Mean $\log(K_p)$ values are presented in Table 21. Of the four, Mn displayed the greatest affinity for the particulate phase (i.e. large value of $\log(K_p)$), whereas Cd and Ni showed the least. These values agree well with those reported in other systems (e.g. McIlroy, 1984).

Estimated partition coefficients were compared with suspended solids concentration to determine whether the negative relationship between the two existed as has been reported by O'Connor and Connolly (1980), McIlroy (1984), Di Toro *et al.* (1985) and others. Results of this analysis were equivocal (Figure 62): whereas Cd appeared to be negatively related to solids concentration in a linear fashion, Ni displayed a

TABLE 21. TRACE METAL PARTITION COEFFICIENTS CALCULATED FROM FIELD SAMPLES

Metal	Log(K_p) (mean $\pm$ 1 SE)
Cd	4.25 $\pm$ 0.04
Ni	4.23 $\pm$ 0.03
Cu	4.77 $\pm$ 0.04
Mn	5.67 $\pm$ 0.04

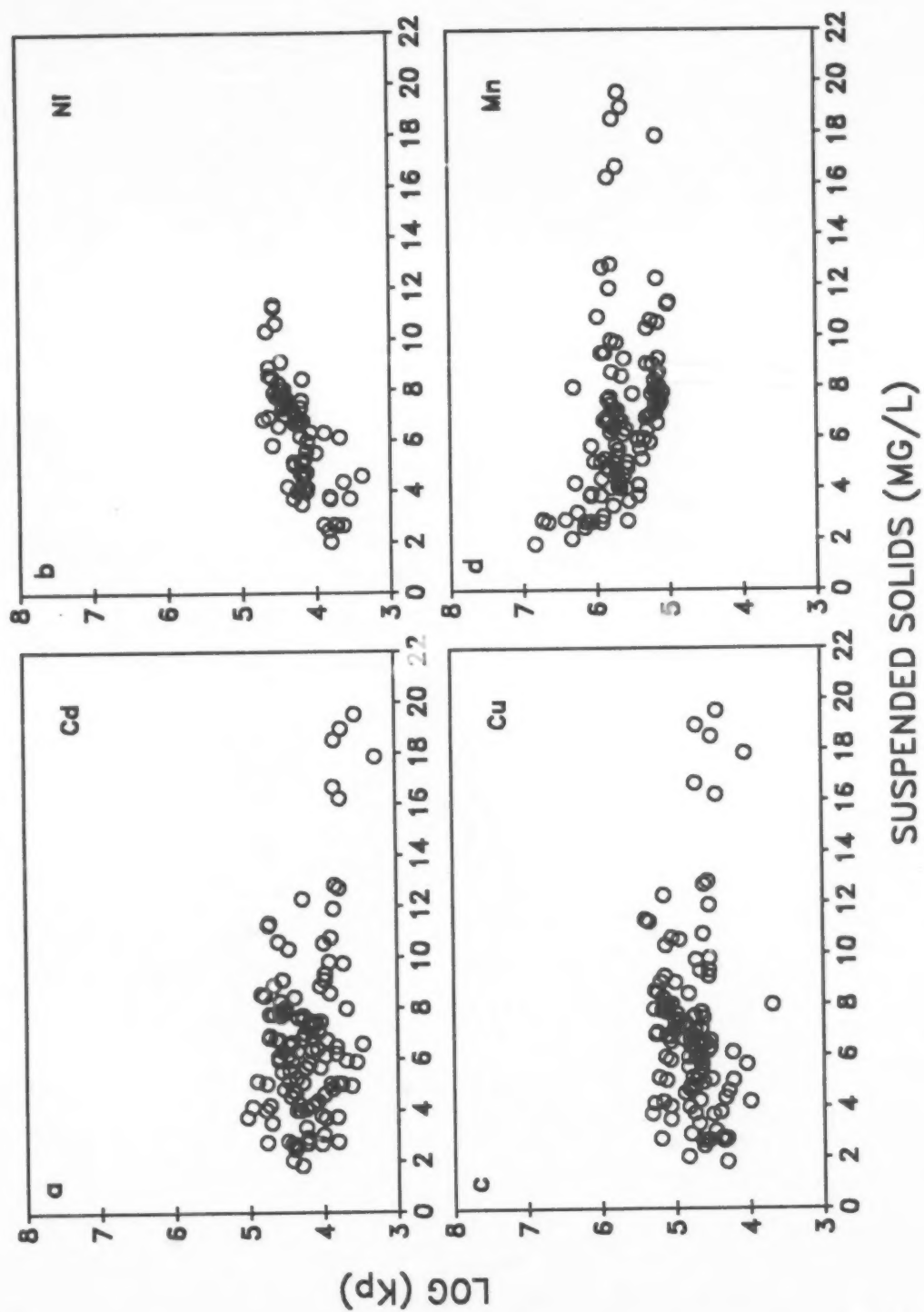


Figure 62. Partition coefficients of cadmium, nickel, copper, and manganese versus suspended solids concentration.

positive relationship, and Cu was unrelated. Mn also showed a negative relationship, although the form of this relationship was distinctly nonlinear. Due to the relatively narrow range of suspended solids encountered during field sampling, it may be that these results are a product of sampling and analysis error. Clear definition of the effect of suspended solids concentration on contaminant phase partitioning in the field awaits further analysis.

Tissue Residues

Mytilus--

Because of their dynamics in the water column, Cd and Pb will be presented as examples of trace metal behavior in Mytilus edulis employed as a biomonitoring tool at the biological effects stations. Patterns of tissue residues differing from those of Cd and Pb will be discussed where they occurred. All trace metal concentrations determined for exposure assessment mussel samples are given in Appendix G. It should be pointed out that statistical comparisons of mean values are not possible due to pseudoreplication of sampling units (see Hurlbert, 1984). Comparisons will therefore be restricted to trends only.

Natural variation in trace metal concentration occurs in mussels which is related to such factors as reproductive and physiological status (e.g. Cossa et al., 1979). Due to the length of this study, mussels deployed at the FVP site were harvested from the Narragansett Bay reference population at different times of the year. Predeployment variation in levels of Cd and Pb in Narragansett Bay mussels can be seen in Figure 63. Levels were highest during the fall and winter months, and lowest during summer. These seasonal trends agree with those seen in mussels elsewhere (Cossa and Rondeau, 1985; but see Goldberg et al., 1978; La Touche and Mix, 1982; Gault et al., 1983). Ranges of all metals are given in Table 22. Differences

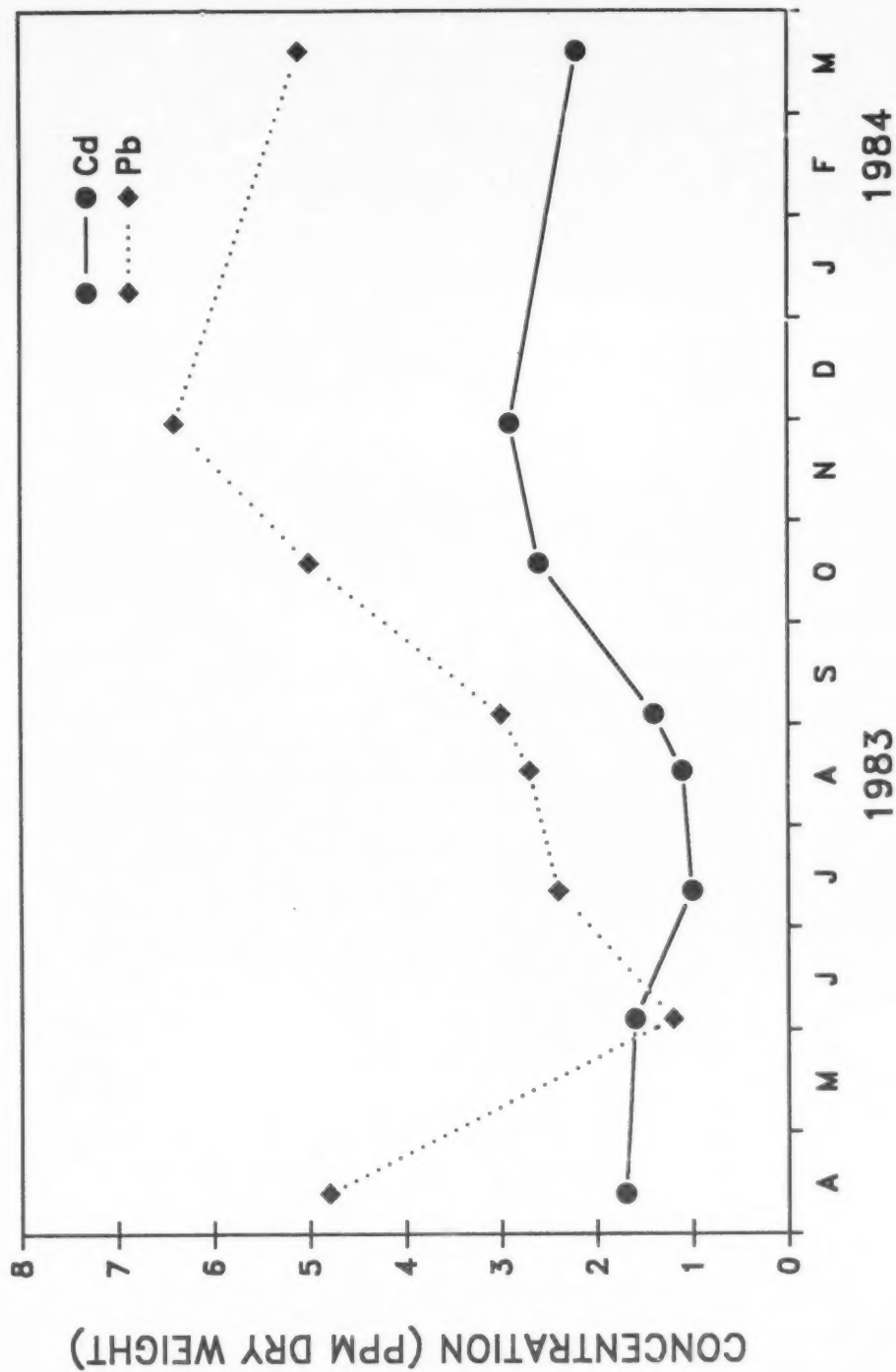


Figure 63. Cadmium and lead tissue residues in Narragansett Bay reference mussels over time.

TABLE 22. RANGE OF TRACE METAL TISSUE RESIDUES IN
NARRAGANSETT BAY STOCK POPULATION MYTILUS

Metal	Range ($\mu\text{g/g}$)
Mn	16 - 74 *
Fe	150 - 896
Ni	1.6 - 7.9
Pb	1.1 - 7.6
Zn	78 - 269
Cr	0.6 - 2.7
Cu	5.7 - 21.8
Cd	1.0 - 3.0

* ranges determined from collections of Narragansett Bay Mytilus on 8 dates

in initial metal concentrations between dates may have influenced spatial and temporal patterns observed in the field, although depuration of elevated levels generally occurred within a few weeks of termination of exposure in laboratory experiments with BRH material (Lake et al., 1985)

Predisposal mussels were retrieved on 22 and 23 April 1983 after 37 days of deployment to determine whether differences associated with stations existed prior to exposure to BRH dredged material. Concentrations of Cd and Pb are shown in Figure 64. Cd levels were nearly identical across stations, and fall within the range of concentrations observed by Greig and Sennefelder (1985) in other areas of CLIS. Although levels of Pb varied somewhat between stations (approximately 35% of an overall mean), no clear differences could be seen between mound stations and REFS. This variation may have resulted from differences in local resuspension and ingestion of sediments (recall Pb's higher affinity for the particulate phase).

Tissue residues in mussels retrieved on 7 June 1983 appeared elevated at 400E relative to those at REFS, as seen for Cd and Pb in Figure 65. (Recall that mussels were not exposed at CTR during the disposal operation). Residues at 1000E also appeared somewhat elevated compared with REFS. However, levels of many metals were actually lower at off-mound stations following disposal than they were in April 1983 (compare Pb levels in Figures 64 and 65). Thus while it would appear that the disposal operation influenced metal tissue residues in Mytilus, temporally varying dynamics in the uptake and retention of trace metals tended to obscure this influence. Temporal variation is seen clearly in Figures 66 (for Cd) and 68 (for Pb), which show Cd and Pb levels measured in mussels deployed throughout 1983. With the exception of Pb concentrations at 400E in early June, the patterns displayed by Cd and Pb in CLIS are strikingly similar in both shape and magnitude to those seen in

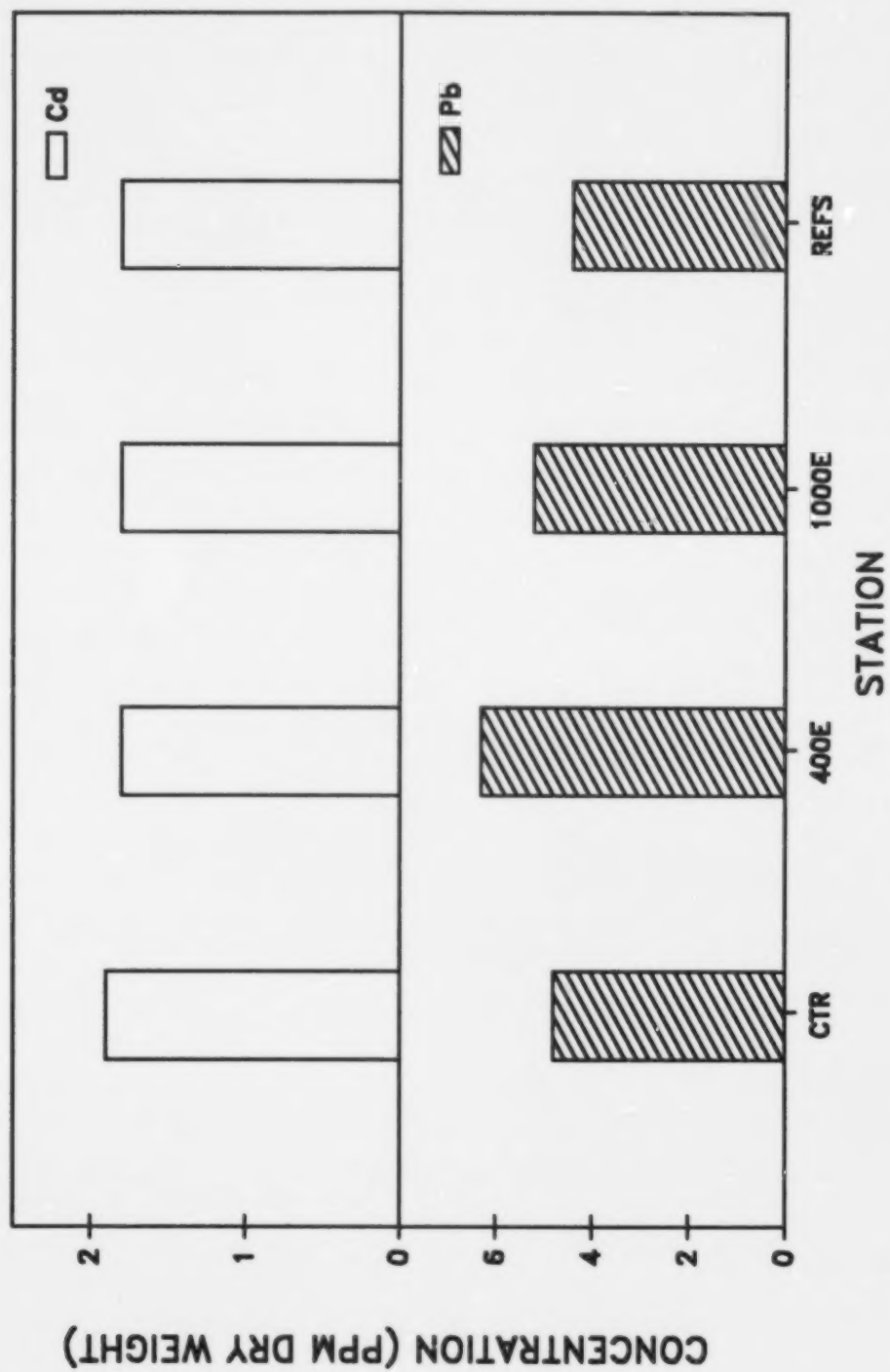


Figure 64. Cadmium and lead tissue residues in predisposal mussels deployed at biological effects stations.

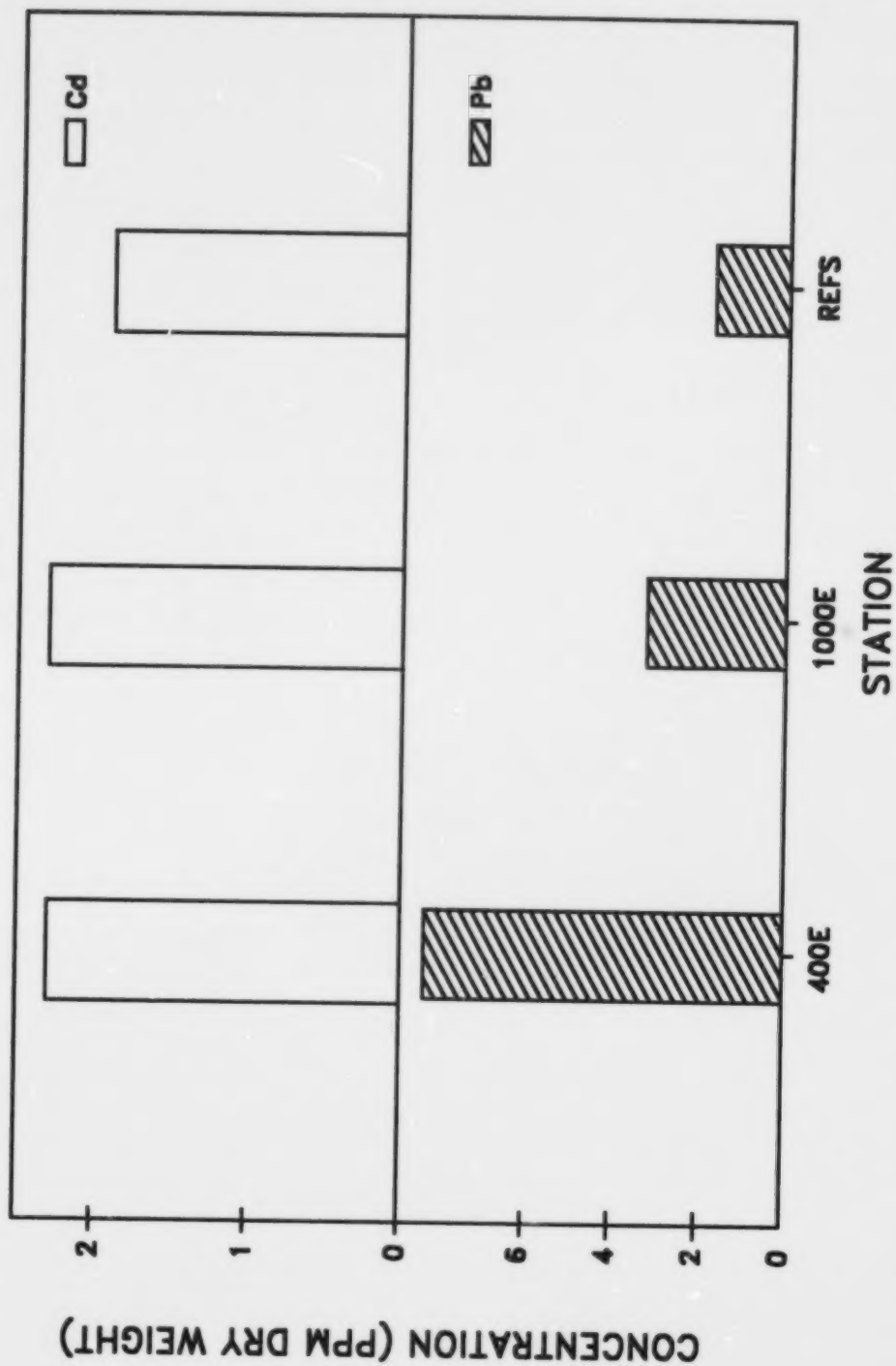


Figure 65. Cadmium and lead tissue residues in postdisposal mussels deployed at biological effects stations.

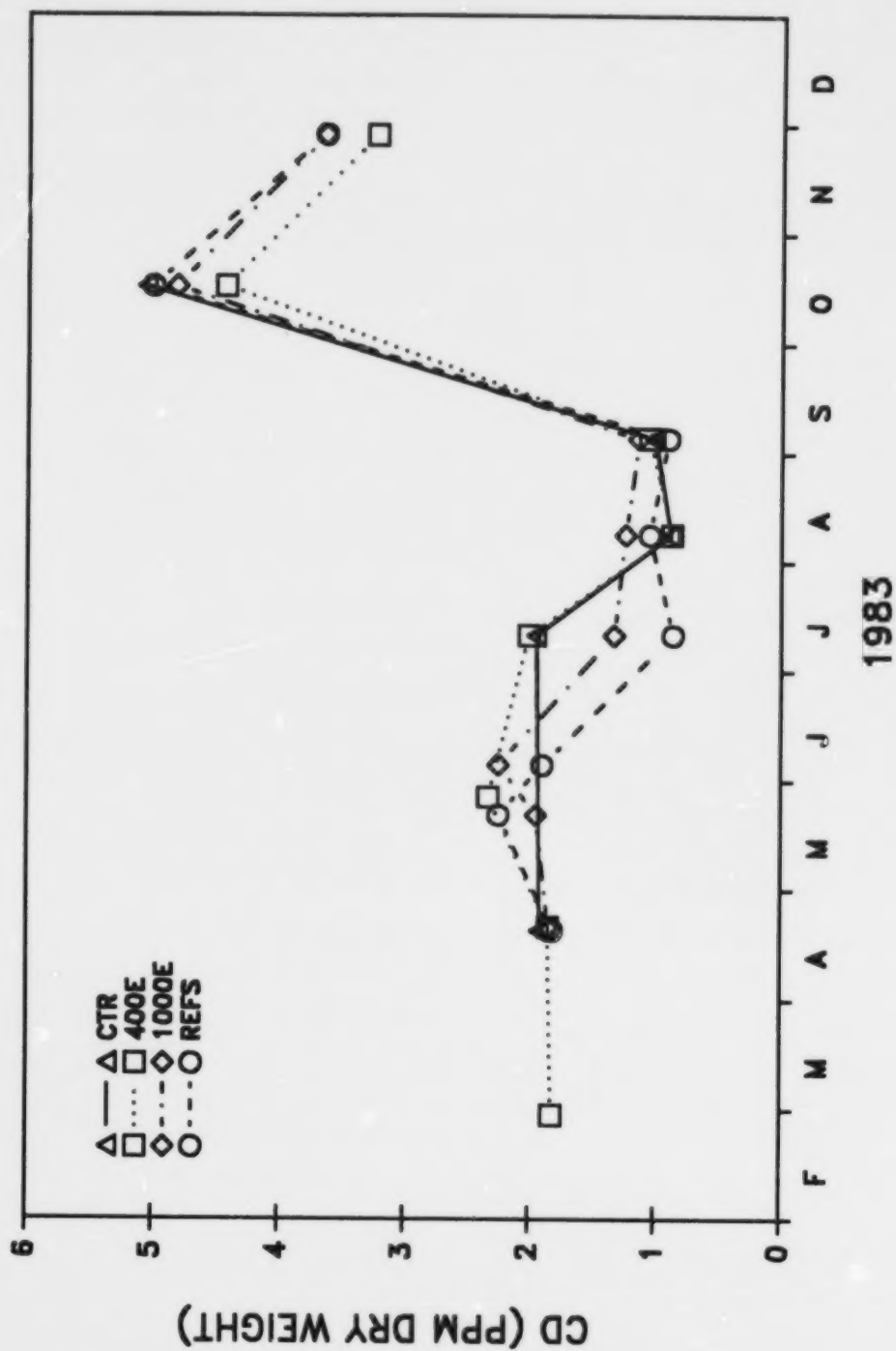


Figure 66. Cadmium tissue residues in mussels deployed at biological effects stations over time.

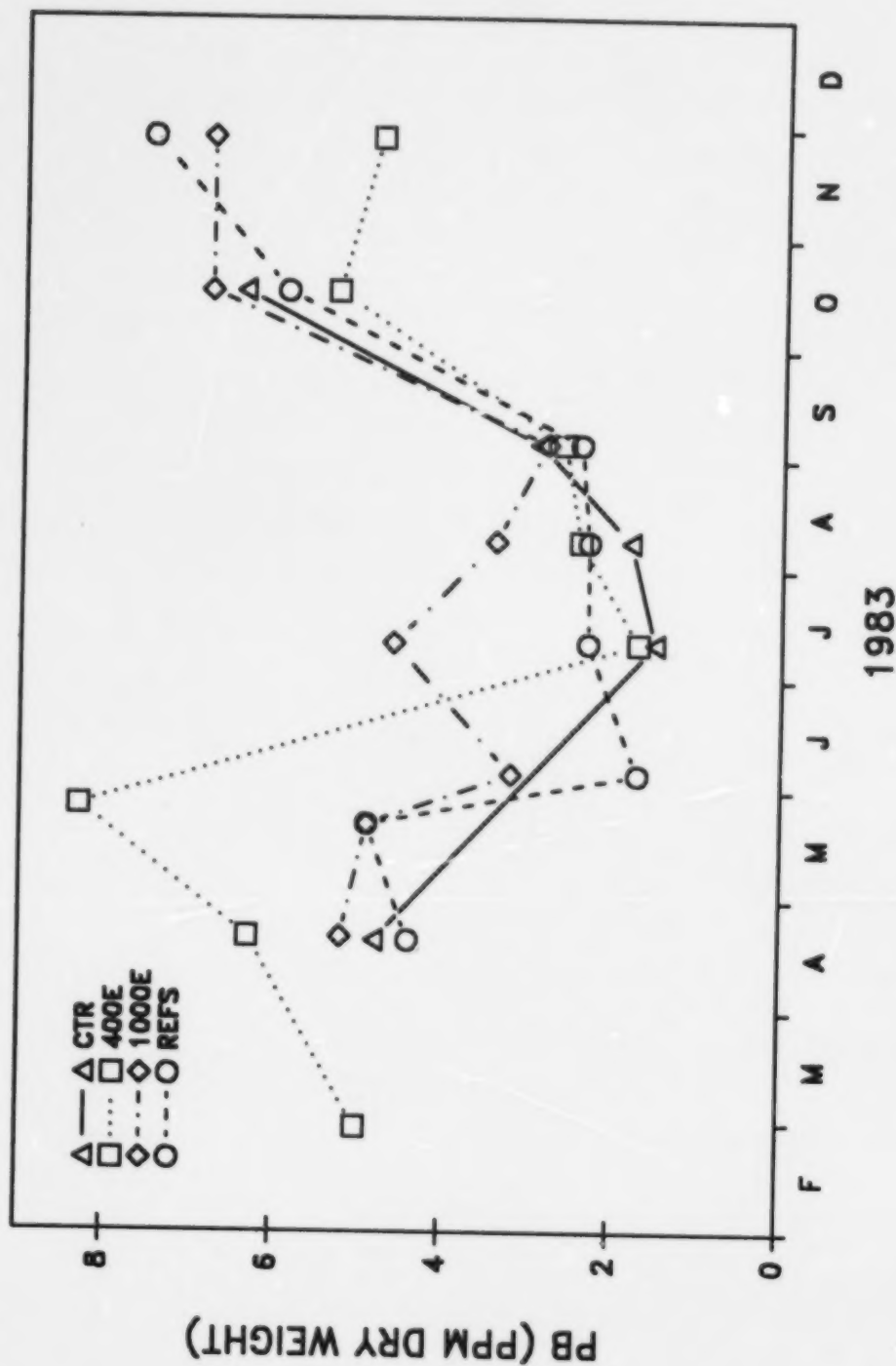


Figure 67. Lead tissue residues in mussels deployed at biological effects stations over time.

Narragansett Bay mussels (Figure 63). The lack of deviation from this pattern by Cd in June 400E may be related to the uptake dynamics of this metal, in conjunction with differences in phase partitioning and other physiochemical processes.

Nephtys--

Trace metal tissue residues in native Nephtys incisa are examined in the following discussion using Cd as an example. Recall that this metal displayed the highest relative elevation in both BRH barrel material and in postdisposal mound sediments. Behavior displayed by metals which is different than that of Cd will be described where appropriate.

Cd levels in Nephtys collected prior to the disposal operation in early March 1983 are given in Figure 68. Despite the relatively large variation displayed by worms at 1000E, these showed Cd concentrations which were significantly higher than those at the other biological effects stations (Duncan's multiple range test, $P < 0.05$). Levels at the remaining stations did not differ statistically. The elevated levels at 1000E are not understood, as predisposal sediment Cd concentrations were not different at this station (see Figures 49 and 54). Zn and Cu also displayed highest levels at 1000E on this date.

Nephtys were not collected from the mound stations immediately subsequent to disposal because resident worms had been buried by dredged material. Due to limited sample sizes obtained from 400E and 1000E (i.e. $n=1$ in both cases), metal tissue residues could not be compared statistically. However, with the exception of Fe, Mn and Cr, all metals tissue residues were found to be higher at 400E, although concentrations were within the range of predisposal levels observed in CLIS.

The temporal pattern of Nephtys Cd tissue residues is shown in Figure 69. No increases attributable to the presence of

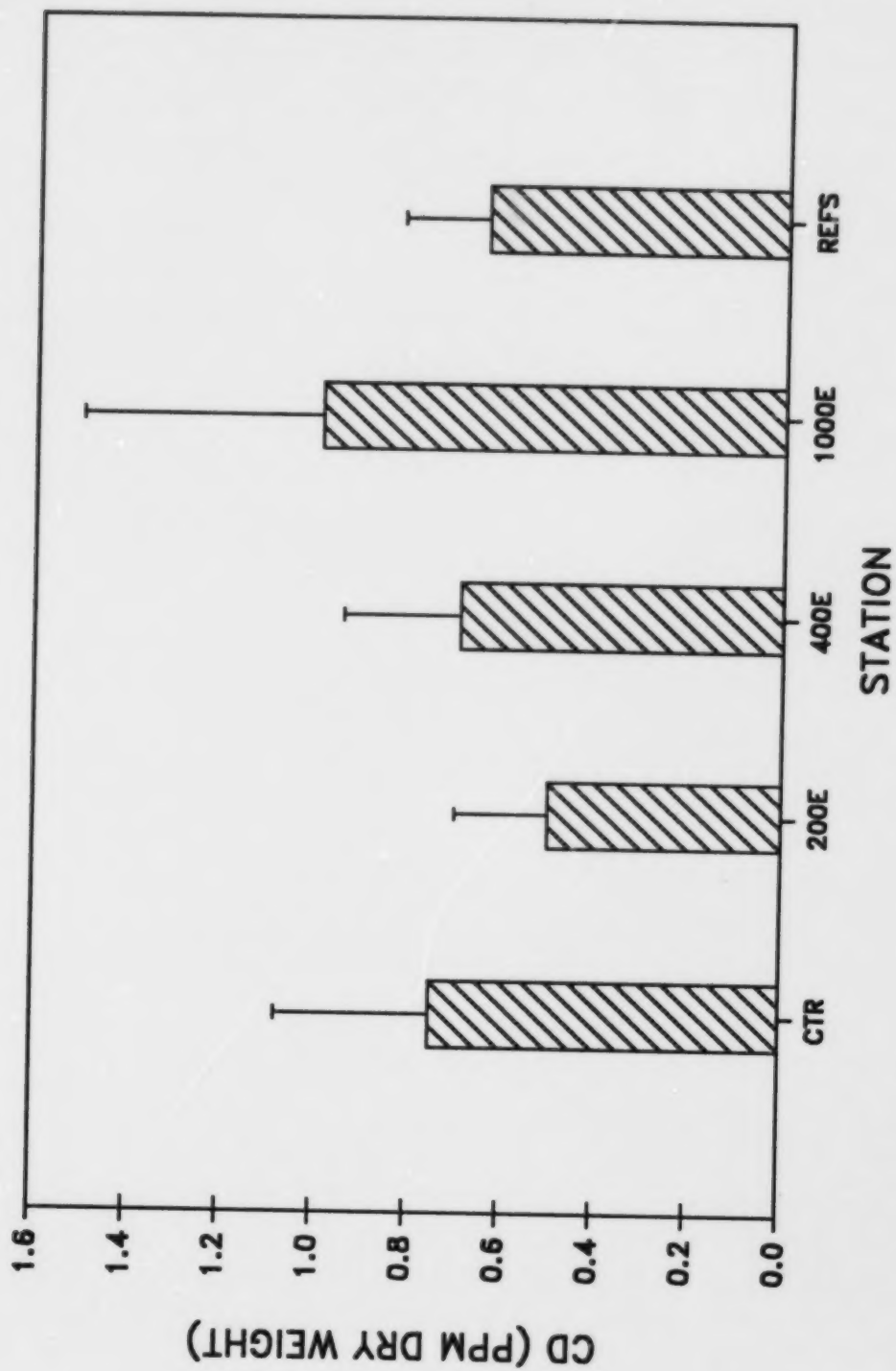


Figure 68. Cadmium tissue residues in predispositional *Nephtys incisa* from biological effects stations.

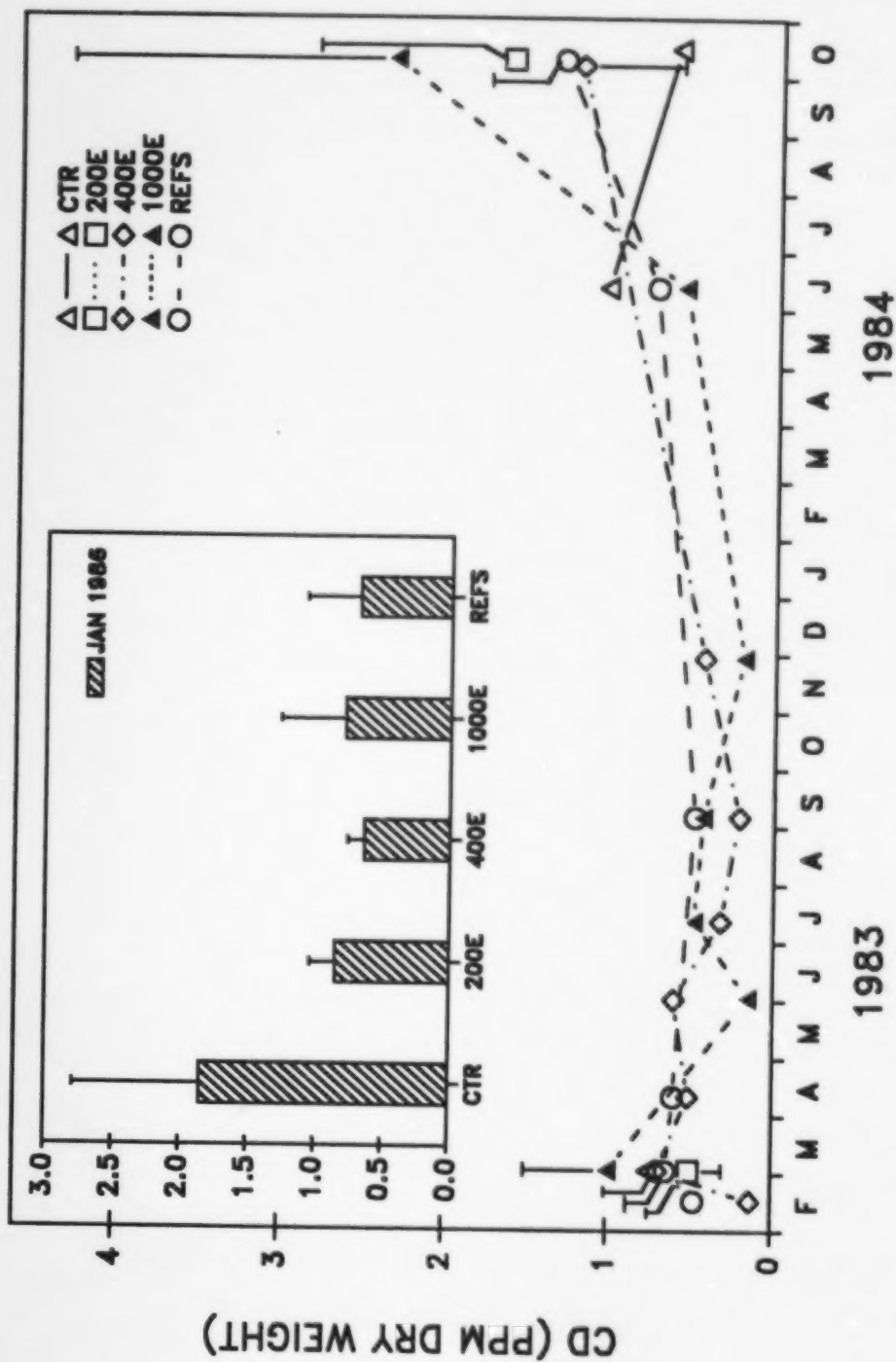


Figure 69. Cadmium tissue residues in Nephtys incisa from biological effects stations over time.

dredged material were seen at 400E, 1000E or REFS in 1983. Although present at relatively high densities in December 1983 (Scott et al., 1987), postdisposal Nephtys samples for trace metal analysis were first collected from CTR in June 1984. Whereas levels of all metals except Fe and Mn were highest at this station, they were generally within the range of predisposal concentrations. The large size of many of the CTR worms suggests that they may have had only recently migrated onto the mound. Only Zn (213 ppm) and Cu (174 ppm) were substantially elevated above background levels. An unexplained increase in both Cd and Cu at all stations but CTR was observed in October 1984. Because the increases were largest at off-mound stations (particularly 1000E), it appears that they were unrelated to contaminants on the disposal mound. However, the low levels at CTR may be a sampling artifact, as only one sample was collected at that station. The inset in Figure 69 shows Cd tissue residues in January 1986; no data were collected between this date and the October 1984 sampling period. Metal concentrations were at background at all stations except CTR, which showed significantly elevated levels of most metals. Although this may reflect slow, long-term uptake of metals from the dredged material, Nephtys at 200E did not show similar elevations.

Summary

Sediments dredged from Black Rock Harbor were highly contaminated with trace metals. Elevations above background CLIS sediments ranged from 1.5 times for Fe to 62 times for Cd. Of eight metals analyzed, only Mn concentrations were lower in the harbor sediments (0.4 times). BRH sediments for laboratory biological effects studies were representative of those in the harbor, despite large variation in metals concentrations among different regions of the study reach.

Prior to disposal of the dredged material in CLIS, trace

metal chemistry of sediments at biological effects stations was similar, and agreed with previous studies in the area. The disposal operation resulted in a gradient of contaminant concentration with highest levels at the center of the mound. Large variation in contaminant concentrations were apparent among different depths of gravity cores collected on the mound. However, mean metal concentrations in surficial and deeper sediments were up to 50% less than those in the barrel material, suggesting loss of much of the contaminant mass to the water column during dredging, barging, and disposal activities. Metal concentrations remained elevated at mound and flank stations throughout the two year sampling program, although surficial concentrations generally decreased following the winter of 1983. This weathering of surficial sediments can be attributed to tidal scouring of fine particulates, oxidative loss, phase partitioning into and subsequent loss of surficial pore waters, and dilution through deposition of resuspended, background sediments. Somewhat surprisingly, elevated metal concentrations were also observed at 1000E, a station previously presumed to be outside of the effects of the disposal operation. Although dredged material may have been transported to this site during disposal or through transport of tidally resuspended mound sediments, evidence exists to suggest the occurrence of an errant dump 50 m to the northwest. In either case, these sediments cannot be ruled out as a potential source of contamination to deployed mussels and native infaunal organisms at that site. Trace metal concentrations in REFS sediments remained low and apparently unaffected by the disposal of BRH material at the FVP site.

Water column chemistry was difficult to characterize throughout the study. Samples collected at CTR over a tidal cycle prior to disposal suggested heterogeneity in contaminant levels between water masses moving through the site, possibly resulting from disposal operations occurring elsewhere in the

CLIS disposal area during this period. Samples were not collected during the FVP disposal operation, so estimates of its effects on water column chemistry could not be made. Evidence of water column contamination from the mound was restricted to near bottom samples obtained at the mound one month following disposal, and appeared to be associated primarily with the particulate phase. Other variation could be explained to a large extent by changes in suspended particulate concentrations. Partition coefficients ranged over 2-3 orders of magnitude between metals, with Cd and Ni displaying the highest affinity for the dissolved phase, and Mn showing the lowest. $\log(K_p)$ appeared to be unrelated to suspended solids concentration for Cu, negatively related for Cd and Mn, and positively related for Ni.

Tissue residues measured in Mytilus edulis deployed at 1 meter above the bottom and retrieved prior to disposal were fairly uniform across stations. Immediately following disposal, levels tended to be higher at 400E and 1000E than at REFS. With the exception of the June 1983 sample from 400E, however, metal residues at all stations behaved similarly to those measured in predeployment, Narragansett Bay reference mussels. Thus it was difficult to distinguish BRH effects from those resulting from physiological changes. Both field-deployed and Narragansett Bay mussels displayed seasonal variation such that tissue residues were highest in winter and lowest in summer.

With the exception of those measured at 1000E, trace metal residues measured predisposal in native Nephtys incisa displayed little difference across stations. Residues of some metals (Cd, Cu, Zn) were found to be highest at 1000E, although sediment concentrations of these elements were not elevated. Levels of most trace metals in worms collected from 400E were slightly elevated relative to those in REFS worms immediately following disposal, but these were within the range of concentrations

measured predisposal. Clear elevations above predisposal conditions were not observed in Nephtys from any station until October 1984. Because levels tended to be highest at 1000E and lowest at CTR, the apparent differences between stations at that time appear to be unrelated to the mound of dredged material.

ORGANIC CHEMISTRY

FVP Study Reach and Barrel Material

Black Rock Harbor and FVP study reach sediments contained high levels of PCBs, and oil and grease. Levels measured in samples collected by NED prior to dredging are given in Appendix E. As with trace metals discussed earlier, large variation in organic contaminant concentration existed among reach stations with lower values occurring near the mouth of the harbor. Mean concentrations of PCBs, and oils and grease measured in the FVP study reach are given in Table 23.

Results of organic analyses of the barrelled BRH sediment used in laboratory experimentation are given Table 24. Abbreviations used for compounds in this table are defined in Table 13. In addition to high levels of PCBs and Ethylan, this material contained high concentrations of both high molecular weight parent PAHs and low molecular weight alkylated PAHs. This indicates that the hydrocarbons originated from both combustion products and fuel oil contamination (Lake et al., 1979). The oil contamination of this sediment was very apparent by both its distinctive odor and the visible sheen the sediment left when mixed with water. Direct comparison of contaminant concentrations found in BRH before dredging (Table 23) with those found in the barrels is only possible for total PCBs, as PAHs were not determined in sediments collected prior to dredging. The concentration of PCBs measured in the barrel material was approximately three times that measure in the FVP

TABLE 23. ORGANIC CONTAMINANT CONCENTRATIONS IN THE FVP STUDY REACH AS DETERMINED BY NED

Contaminant	Mean	Standard error	Coefficient of variation (%)
PCBs (as Aroclor® 1254)	2,130 ppb	470	106
Oil and Grease	17,700 ppm	1,980	56

TABLE 24. ORGANIC CONTAMINANT CONCENTRATIONS IN BRH BARREL MATERIAL

Compound	Mean (ng/g dry)	Standard error	Coefficient of variation (%)
P0S166*	816	126	34.6
H1C166	1,780	196	24.7
H2C166	2,597	303	26.1
H3C166	2,886	289	22.4
H4C166	2,440	229	21.0
H0S166	9,703	1,016	23.4
P1A178	5,220	724	31.0
P2B178	1,718	228	29.6
P0S178	6,938	946	30.5
H1C178	8,274	953	25.8
H2C178	9,335	1,103	26.4
H3C178	7,297	908	27.8
H4C178	4,503	775	38.5
H0S178	29,409	3,728	28.3
P0S184	352	55.9	35.5
H1C184	1,333	181	30.3
H2C184	2,780	346	27.8
H3C184	2,787	352	28.3
H4C184	1,769	204	11.5
H0S184	8,669	1,066	27.5
P1A202	6,468	515	17.8
P2B202	7,890	471	13.3
P0S202	15,261	1,016	14.9
H1C202	7,440	538	16.2
H2C202	4,479	258	12.9
H3C202	3,227	204	14.2
H4C202	1,418	114	17.9
H0S202	16,564	1,085	14.6
P1A228	3,997	271	15.2
P2B228	5,255	342	14.6
P0S228	10,389	812	17.5
H1C228	5,866	343	13.1
H2C228	3,787	222	13.1
H3C228	2,184	116	11.9
H4C228	892	55.9	14.0
H0S228	12,729	706	12.4

(continued)

TABLE 24 (continued).

Compound	Mean (ng/g dry)	Standard error	Coefficient of variation (%)
P1A252	6,950	443	14.3
P2B252	3,370	236	15.6
P3B252	3,977	393	22.1
P4B252	799	89.4	25.0
P0S252	15,097	1,152	17.1
H1C252	5,394	340	14.1
H2C252	2,715	222	18.3
H3C252	1,190	99.7	18.7
H4C252	342	30.4	19.9
H0S252	9,642	678	15.7
P0S276	9,292	995	24.0
P0S278	5,291	808	34.1
P0S300	855	68.0	17.8
P0S302	4,385	260	13.3
Ethylan	3,887	322	18.5
PCBs	6,490	372	12.8

- * PA1nnn = parent PAH A with molecular weight nnn
 P0Snnn = sum of parent PAHs with molecular weight nnn
 H1Cnnn = alkyl homolog 1 of parent PAH with molecular weight nnn
 H0Snnn = sum of alkyl homologs of parent PAH with molecular weight nnn
 See Table 13 for compound names

study reach. Although both NED and ERLN quantified PCBs as Aroclor® 1254, this discrepancy may be due to differences in the analytical techniques used. It appears unlikely that PCBs were accidentally introduced between barrel material collection and analysis. PCB concentrations within the reach varied from non-detectable levels to 9,170 ppb, compared to an average of $6,490 \pm 370$ ppb from the barrels. Thus organic contaminant levels in the barrels were within the range of concentrations in the study reach, but that range is so broad as to make the comparison not particularly useful.

The results listed in Table 24 indicate the degree of homogeneity achieved with the mixing procedure. The coefficient of variation of the group of hydrocarbons having the greatest concentration, the alkylated phenanthrenes and anthracenes (parent molecular weight 178), was 38.5%, while that of the PCBs was 12.8%. These deviations are similar to those found for trace metals (Table 18) and with a previous in-house reference material, a mussel homogenate used for quality control purposes (Galloway *et al.*, 1983).

In addition to individual contaminant concentrations, two types of information could be gleaned from the PAH data: the absolute quantity of aromatic hydrocarbons of interest, and the relative distribution of this total among the various compounds (the 'quality' of hydrocarbon contamination). Quantitative information was readily abstracted by summing the concentrations of individual compounds into two classes, parents and homologs, or simply a total aromatic hydrocarbon sum. These may be expressed as:

$$\text{PSUM} = \text{POS166} + \text{POS178} + \text{POS202} + \text{POS228} + \text{POS252} + \\ \text{POS276} + \text{POS278} + \text{POS300} + \text{POS302},$$

$$\text{HSUM} = \text{HOS166} + \text{HOS178} + \text{HOS202} + \text{HOS228} + \text{HOS252}, \text{ and}$$

$$\text{SUM} = \text{PSUM} + \text{HSUM},$$

where PSUM is the sum of the parent PAHs measured, HSUM is the sum of the alkylated homologs measured, and SUM is the total sum of all PAHs measured. Note that the dibenzothiophenes (parent molecular weight 184) were not included in these summations because they are not strictly aromatic hydrocarbons. The statistic SUM captures most of the quantitative information from the samples.

Qualitative information was somewhat more difficult to present. Figure 70 represents the distribution of aromatic compounds measured in the barrel samples, while Figure 71 represents the distribution in sediments collected from REFS. There is a clear difference in the patterns displayed in these two figures. Figure 70 shows a generally even distribution of contaminants from low to high molecular weight. This is due to the presence of high concentrations of both fuel oil-derived aromatics with low molecular weight and combustion-derived aromatics with high molecular weight. In contrast, Figure 71 shows a distribution of aromatics strongly skewed towards high molecular weight compounds, that is, combustion-derived aromatics with little evidence of fuel oil contamination. Centroid statistics, with units of molecular weight, were chosen to represent these qualitative differences. Their values were calculated as intensity-weighted averages of all measured aromatic hydrocarbons in the following way:

$$\text{PCENT} = (\text{POS166} \cdot 166 + \text{POS178} \cdot 178 + \text{POS202} \cdot 202 + \text{POS228} \cdot 228 + \text{POS252} \cdot 252 + \text{POS276} \cdot 276 + \text{POS278} \cdot 278 + \text{POS300} \cdot 300 + \text{POS302} \cdot 302) / \text{PSUM},$$

$$\text{HCENT} = (\text{H1C166} \cdot 180 + \text{H2C166} \cdot 194 + \text{H3C166} \cdot 208 + \text{H4C166} \cdot 222 + \text{H1C178} \cdot 192 + \text{H2C178} \cdot 206 + \text{H3C178} \cdot 220 + \text{H4C178} \cdot 234 + \text{H1C202} \cdot 216 + \text{H2C202} \cdot 230 + \text{H3C202} \cdot 244 + \text{H4C202} \cdot 258 +$$

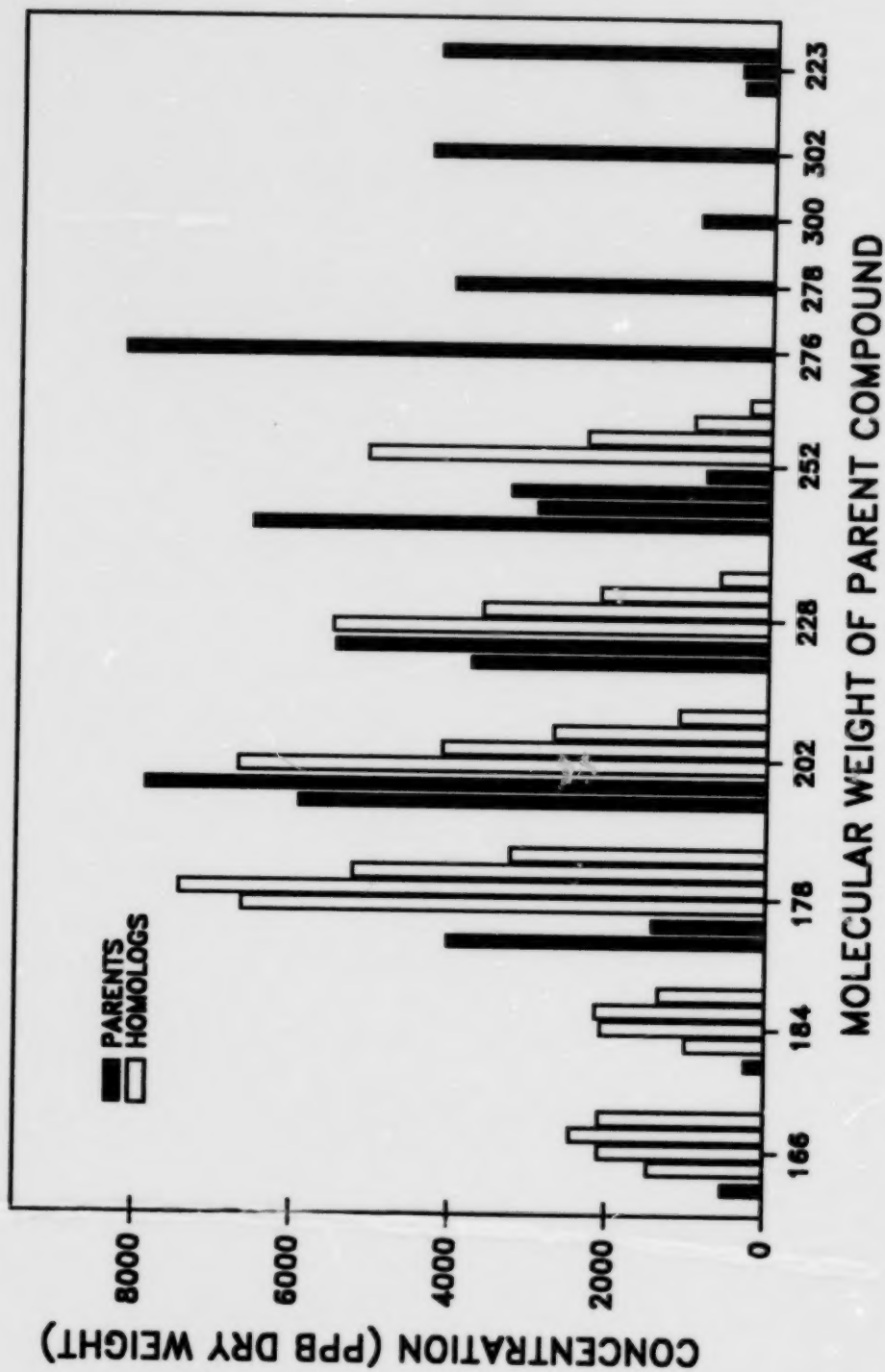


Figure 70. Molecular weight distribution of PAH compounds in BRH barrel material.

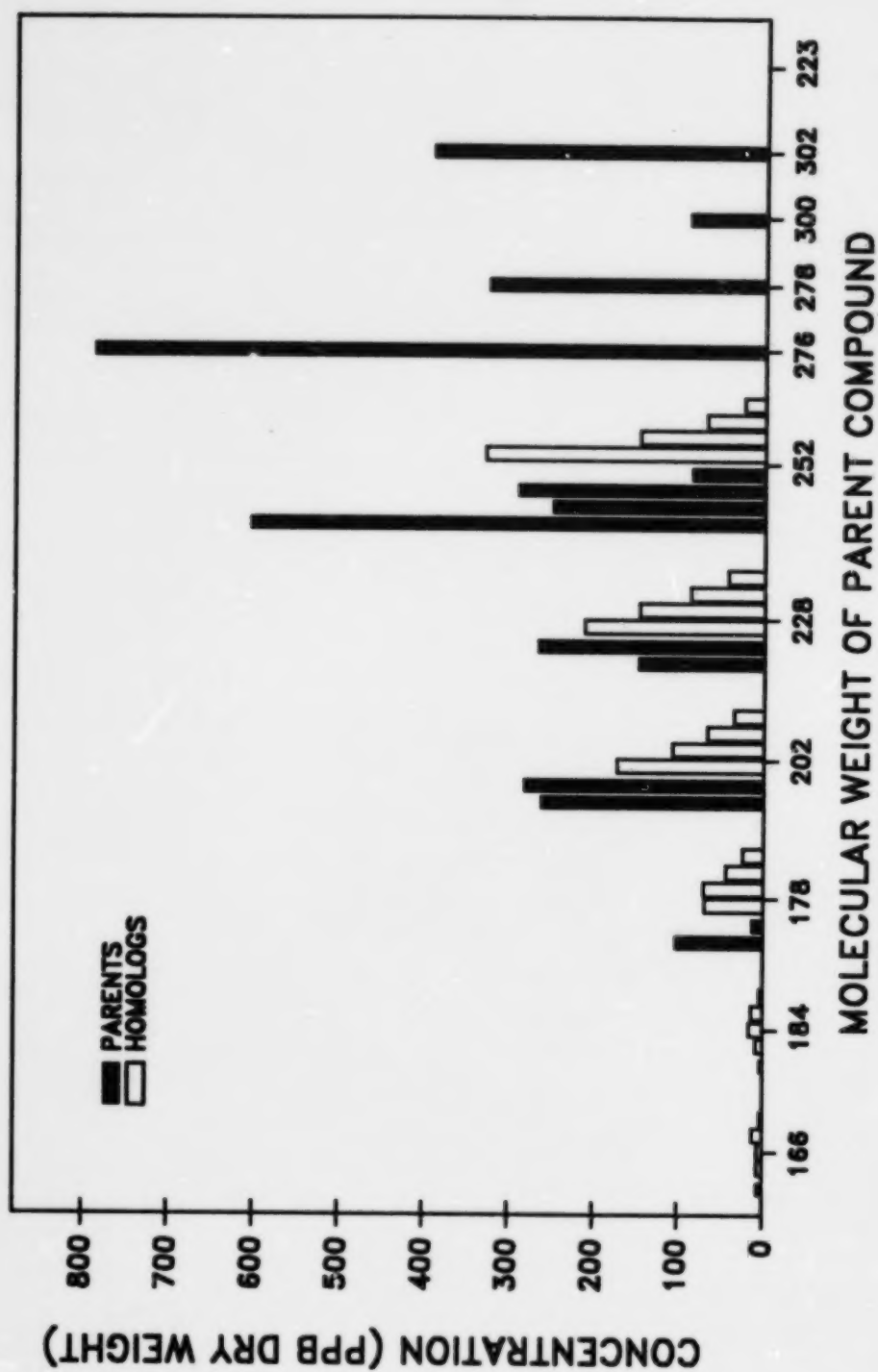


Figure 71. Molecular weight distribution of PAH compounds in REFS sediments.

$$\begin{aligned} & \text{H1C228*242} + \text{H2C228*256} + \text{H3C228*270} + \text{H4C228*284} + \\ & \text{H1C252*266} + \text{H2C252*280} + \text{H3C252*294} + \\ & \text{H4C252*308})/\text{HSUM}, \text{ and} \end{aligned}$$

$$\text{CENT} = (\text{PCENT*PSUM} + \text{HCENT*HSUM})/\text{SUM},$$

where PCENT is the centroid of the PAH parent compounds, HCENT is the centroid of the alkyl homolog compounds, and CENT is the overall centroid of all measured PAHs. In the following discussion of organic chemistry, the statistics SUM and CENT will be presented to describe the dynamics of PAHs. Information concerning single hydrocarbons will be presented when the behavior of individual compounds differed from that of the summary statistics. As Ethylan and PCB concentrations were not necessarily related to any particular hydrocarbon concentration, data describing their chemistry will also be presented. Individual organic compound concentrations are given in Appendix H. Mean values of the two summary statistics, PCBs, and Ethylan measured in barrel material are given in Table 25.

Sediment Organic Chemistry

Organic contaminant levels found in sediment samples collected prior to the disposal operation are given in Table 26. Predisposal samples from 1000E were not analyzed and do not appear in this table. Although not tested statistically due to lack of replication at three of the four stations, there appears to be little difference between the levels for CTR, 200E, and 400E. Concentrations of organic contaminants from REFS, however, appear slightly lower than those from the other three stations. In addition, the value of CENT was somewhat higher, suggesting low level contamination of the mound stations by lower molecular PAHs prior to disposal. This contamination, if real (statistical tests were not performed), may have resulted from historic disposal operations in the CLIS disposal area. The pesticide

TABLE 25. SUMMARY STATISTICS, PCBs, AND ETHYLAN IN BRH BARREL MATERIAL

Compound	Mean	Standard error	Coefficient of variation (%)
SUM*	146,400	13,040	19.9
CENT†	232.8	0.55	0.5
PCBs*	6,490	372	12.8
Ethylan*	3,887	321.8	18.5

* concentrations reported as ng/g dry weight

† units of molecular weight

TABLE 26. MEAN (SE) PREDISPOSAL SUMMARY STATISTICS
PCBS AND ETHYLAN CONCENTRATIONS IN BIOLOGICAL
EFFECTS STATION SEDIMENTS

Contaminant	CTR	Station		REFS
		200E	400E	
SUM*	5,090	4,930	5,880	4,000 (345)†
CENT ^S	247.6	248.9	247.7	249.2 (1.4)
PCBs*	76.5	74.6	97.5	63.9 (2.0)
Ethylan*	0	0	0	0

* concentrations reported as ng/g dry weight

† n=3 for REFS, n=1 for all other stations

^S units of molecular weight

Ethylan was not found in predisposal sediments at any station.

Ratios of organic contaminants in BRH barrel material to those in predisposal sediments are presented in Table 27. PCB and PAH concentrations were generally between 25 and 85 times higher in the barrel material. The relative enrichment by lower molecular weight PAHs of the barrel material is reflected by lower value of CENT. The contaminant ratio of infinity for Ethylan reflects the lack of detectable levels of that compound in CLIS background sediments. Contamination of field sediments by dredged material could therefore be detected by elevated levels of PCBs, increased values for SUM, decreased values for CENT, and the presence of Ethylan.

The disposal operation resulted in large elevations in surficial organic contaminants, as shown in Figure 72. Elevations were greatest at CTR and 200E, and somewhat smaller at 400E. The value of CENT was depressed at these three stations as well. As with the trace metal chemistry discussed in the previous section, evidence of contamination was seen at 1000E in the form of increased Ethylan concentrations (2.6 ± 2.6 ppb). Somewhat decreased CENT values were also measured at that station. However, concentrations of PAHs and PCBs did not appear to differ from those measured at REFS (predisposal 1000E organic levels were not obtained).

Levels of organic contaminants measured following disposal were 0.5 to 0.1 times those measured in BRH barrel material (see Table 27), as was observed for trace metal concentrations. Thus a large portion of the total contaminant mass was apparently lost during the dredging, barging, and disposal operations. Some loss may be attributable to postdisposal scouring, however, as surficial concentrations measured on 26 July 1983 were somewhat lower than those measured in deeper sediments (see Appendix H).

TABLE 27. ORGANIC CONTAMINANT RATIOS OF BRH BARREL MATERIAL
VERSUS PREDISPOSAL SURFICIAL SEDIMENTS

Contaminant	BRH barrels	CLIS predisposal	Ratio
SUM*	146,400	4,970	29.5
CENT†	232.8	248.4	0.94
PCBs*	6,490	78.1	83.1
Ethylan*	3,887	0	∞

* concentrations reported as ng/g dry weight

† units of molecular weight

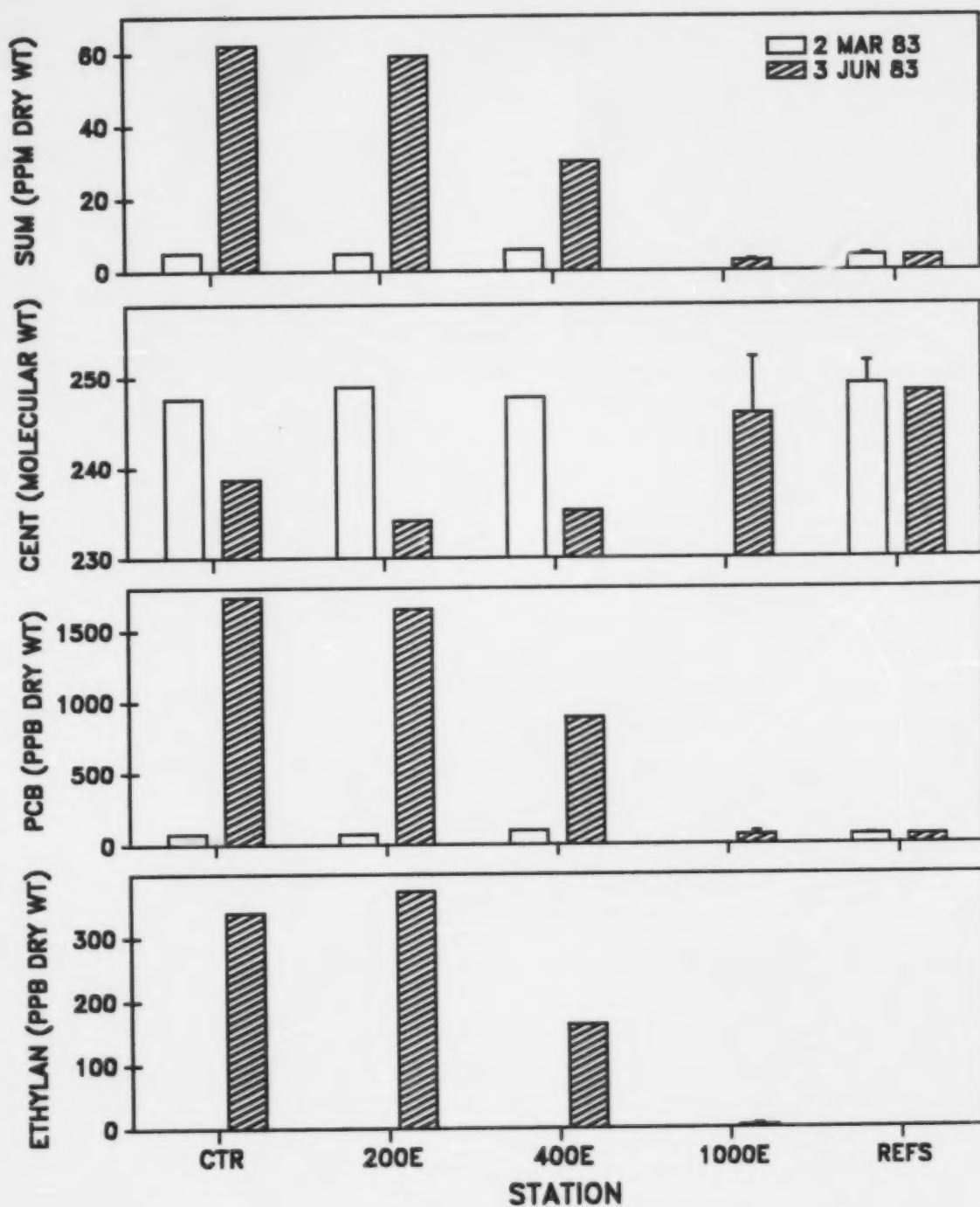


Figure 72. Organic contaminants in predisposal and postdisposal biological effects sediments.

Although more variable than trace metal concentrations, levels of organic contaminants at REFS appeared unaffected by the disposal operation. Figure 73 gives organic concentrations measured in various depths at that station over the course of the study. No Ethylan was detected at REFS at any time. Values of other organic contaminants were slightly lower than those measured predisposal at the remaining biological effects stations (see Table 26), as noted earlier in the discussion of predisposal concentrations.

Elevated surficial levels observed on mound and flank stations following the disposal operation (Figure 74) decreased over time, such that by March 1984, concentrations were approaching those seen at REFS. Strong signals at CTR, 200E and 400E were still observed in September 1984, however, and were particularly noticeable in the PAH summary statistic CENT. This measure remained several molecular weight units lower than that measured at REFS, indicating a relative abundance of low molecular weight hydrocarbons. Ethylan was also present at that time. The processes involved in this weathering phenomenon are not completely understood, although solubilization of compounds and dilution of surficial sediments were likely involved. However, simple solubility does not seem to be the dominant mechanism because Ethylan, a relatively soluble contaminant, remained elevated on the mound over the course of the study.

Water Column Organic Chemistry

Trends in Time and Space--

Organic contaminant concentrations measured in the water column at CTR and REFS prior to disposal are given in Table 28. Because whole water samples were not collected near the bottom at either station, total concentrations are estimated in this table as the sum of those measured in dissolved and particulate phases, whereas CENT was estimated by weighting the dissolved

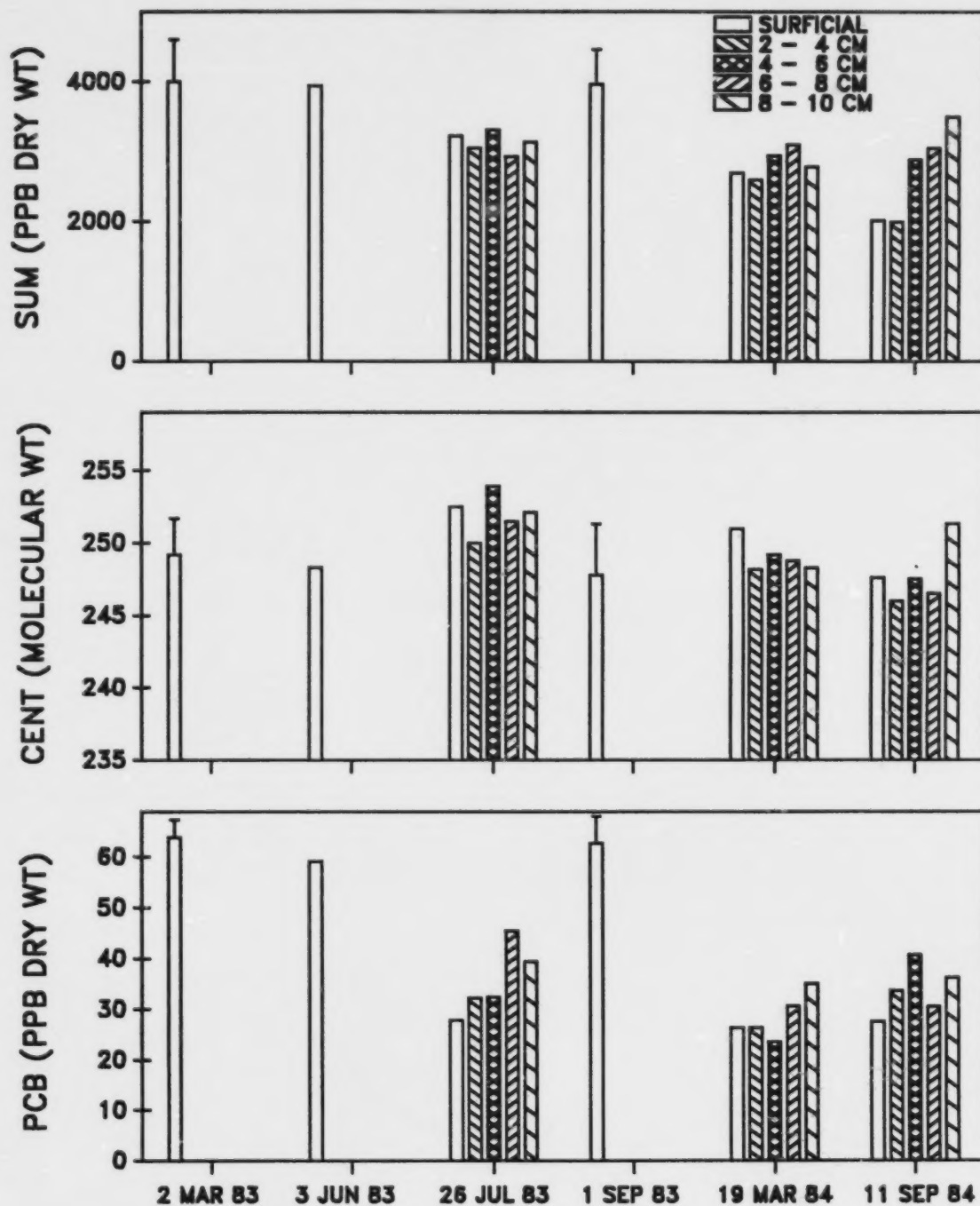


Figure 73. Depth profiles of organic contaminants in REFS sediments over time.

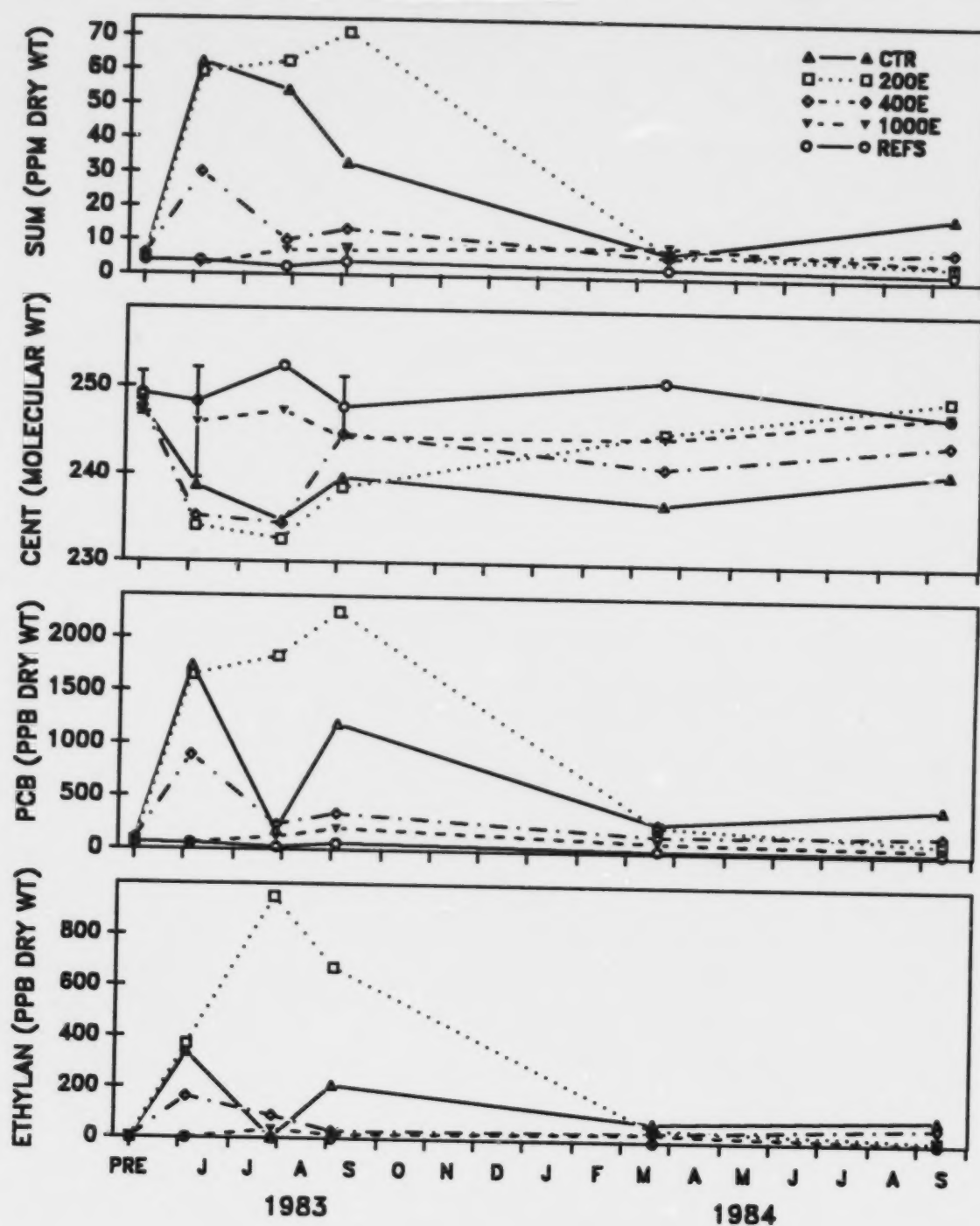


Figure 74. Organic contaminants in biological effects station surficial sediments over time.

TABLE 28. PREDISPOSAL WHOLE WATER ORGANIC
CONTAMINANT CONCENTRATIONS

Contaminant	Depth	Station	
		CTR	REFS
SUM*	surface	8.39	8.67
	bottom†	5.86	2.17
CENT [§]	surface	251.0	242.7
	bottom††	247.1	251.1
PCBs*	surface	0.31	0.39
	bottom††	0.53	0.35
Ethylan*	surface	0	0
	bottom††	0	0

* concentrations reported as ng/kg

† values estimated as the sum of particulate and dissolved phases

§ units of molecular weight

†† values estimated as the weighted mean of sum of particulate and dissolved phases

and particulate values by the concentration of PAHs in the two phases. Although not tested statistically due to lack of replication, surface (5 MBS) samples contained somewhat higher levels of total PAHs at both stations than did bottom waters. The differences between depths were not great however. No large differences in individual contaminant concentrations were seen between stations. As observed in predisposal sediments, no Ethylan was detected in the water column prior to disposal.

The influence of the disposal operation on organic water column chemistry can be seen in Figure 75. Levels of PAHs, PCBs, and Ethylan measured in bottom waters on 7 June 1983 all displayed order-of-magnitude elevations above predisposal concentrations at CTR. Surface waters at this station may also have been affected, as levels of PAHs (23.5 ng/kg) and PCBs (0.7 ng/kg) were two to three times those seen predisposal (see Table 28). Some contamination was also evident in bottom waters at REFS in the form of increased PAH and PCB concentrations, although these were not elevated to the same extent as was seen at CTR.

The organic contaminant signal in near bottom waters persisted through early September 1983 in the form of increased PCB levels relative to those at REFS. However, concentrations of all contaminants had fallen substantially by 21 July. During this period, PAHs consisted to a large extent of lower molecular weight compounds, as indicated by the summary statistic CENT. Concentrations of PCBs and the value of CENT also indicated low level contamination at 400E and 1000E when samples were first collected at these stations in early September. Organic contaminant levels had generally returned to predisposal levels by December 1983.

Phase Partitioning--

Partition coefficients estimated from organic contaminant

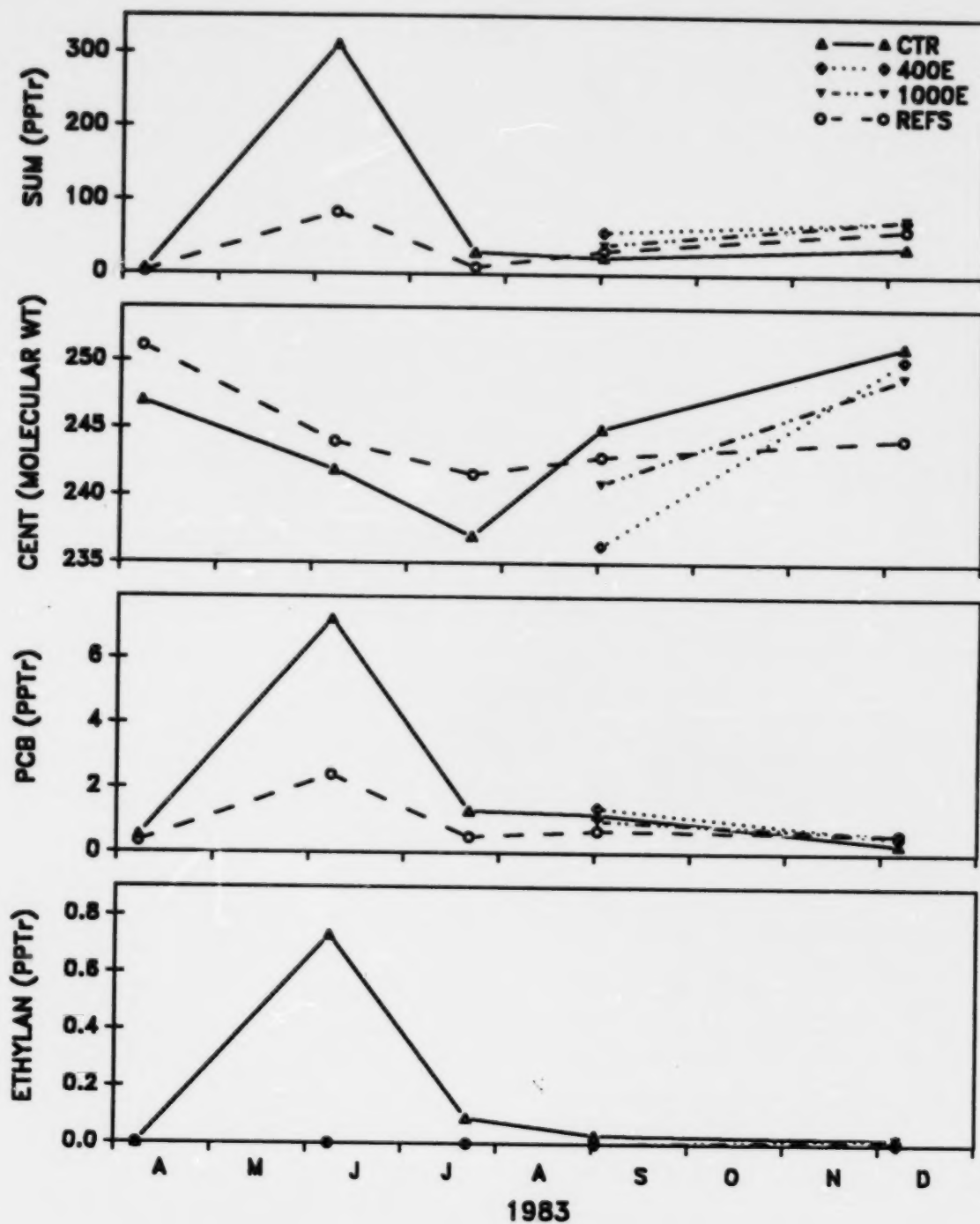


Figure 75. Organic contaminants in near-bottom whole water samples from biological effects stations over time.

concentrations in the field are given in Table 29. Because partition coefficients are contaminant-specific, values for SUM and CENT were not calculated. $\log(K_p)$ for individual PAH compounds ranged between 4.0 and 7.5, in agreement with values reported in the literature (e.g. Pruell *et al.*, 1986). Solubilities of the various compounds were reflected in a generally increasing trend of $\log(K_p)$ with molecular weight, as displayed graphically in Figure 76. Superimposed on this relationship is variation resulting from solubility differences related to molecular structure. Estimated hydrocarbon partition coefficients displayed no discernible trends in time or space, suggesting the partitioning dynamics to have occurred fairly rapidly after introduction to the water column.

Di Toro (in press) has proposed a model relating $\log(K_p)$ of environmental contaminants to suspended solids concentration, and has summarized PCB partition coefficients from a large number of aquatic systems. The partition coefficient estimated for PCBs in this study (5.36 l/kg, Table 29) agrees well with those determined in a similar range of suspended solids (10^0 - 10^1 mg/l) in fresh water and marine systems. PCB $\log(K_p)$ was unrelated to suspended solids concentration over the narrow range of solids concentrations observed during organic sampling (Figure 77).

Tissue Residues

Mytilus--

Organic contaminant concentrations in the Narragansett Bay reference population of Mytilus were examined to determine whether seasonal changes in tissue residues occurred. Although data are available only for April through November 1983, no clear differences were observed between seasons (Figure 78). Relatively low levels of PAHs and PCBs were observed throughout this period. Results similar to these were observed by

TABLE 29. ORGANIC CONTAMINANT PARTITION
COEFFICIENTS CALCULATED FROM FIELD
SAMPLES

Compound	N	Log(K_p)	Standard error
P0S166*	5	4.66	0.27
H1C166	4	4.61	0.24
H2C166	4	4.56	0.18
H3C166	4	4.59	0.09
H4C166	4	4.55	0.17
P1A178	6	5.12	0.25
P2B178	4	5.12	0.23
H1C178	6	5.07	0.22
H2C178	6	5.15	0.11
H3C178	6	5.18	0.07
H4C178	5	5.33	0.12
P0S184	1	4.11	--
H1C184	2	4.29	0.01
H2C184	3	4.65	0.08
H3C184	3	4.85	0.15
H4C184	3	5.08	0.40
P1A202	6	5.44	0.16
P2B202	6	5.41	0.17
H1C202	6	5.70	0.29
H2C202	6	5.97	0.21
H3C202	4	6.15	0.26
H4C202	3	6.04	0.20
P1A228	5	5.92	0.17
P2B228	6	5.91	0.14
H1C228	5	6.14	0.12
H2C228	4	6.57	0.26
H3C228	3	6.34	0.15
H4C228	1	5.06	--
P1A252	5	6.34	0.08
P2B252	4	6.27	0.10
P3B252	5	6.57	0.17
P4B252	4	6.71	0.13
H1C252	2	6.29	0.23
H2C252	1	7.14	--
H3C252	0	--	--
H4C252	0	--	--

(continued)

TABLE 29 (continued).

Compound	N	Log(K_p)	Standard error
POS276	4	7.40	0.25
POS278	2	6.88	0.57
POS300	1	6.23	--
POS302	1	7.39	--
Ethylan	2	5.39	0.33
PCBs	22	5.36	0.05

- * PAInnn = parent PAH A with molecular weight nnn
 POSnnn = sum of parent PAHs with molecular weight nnn
 HlCnnn = alkyl homolog 1 of parent PAH with molecular weight nnn
 See Table 13 for compound names

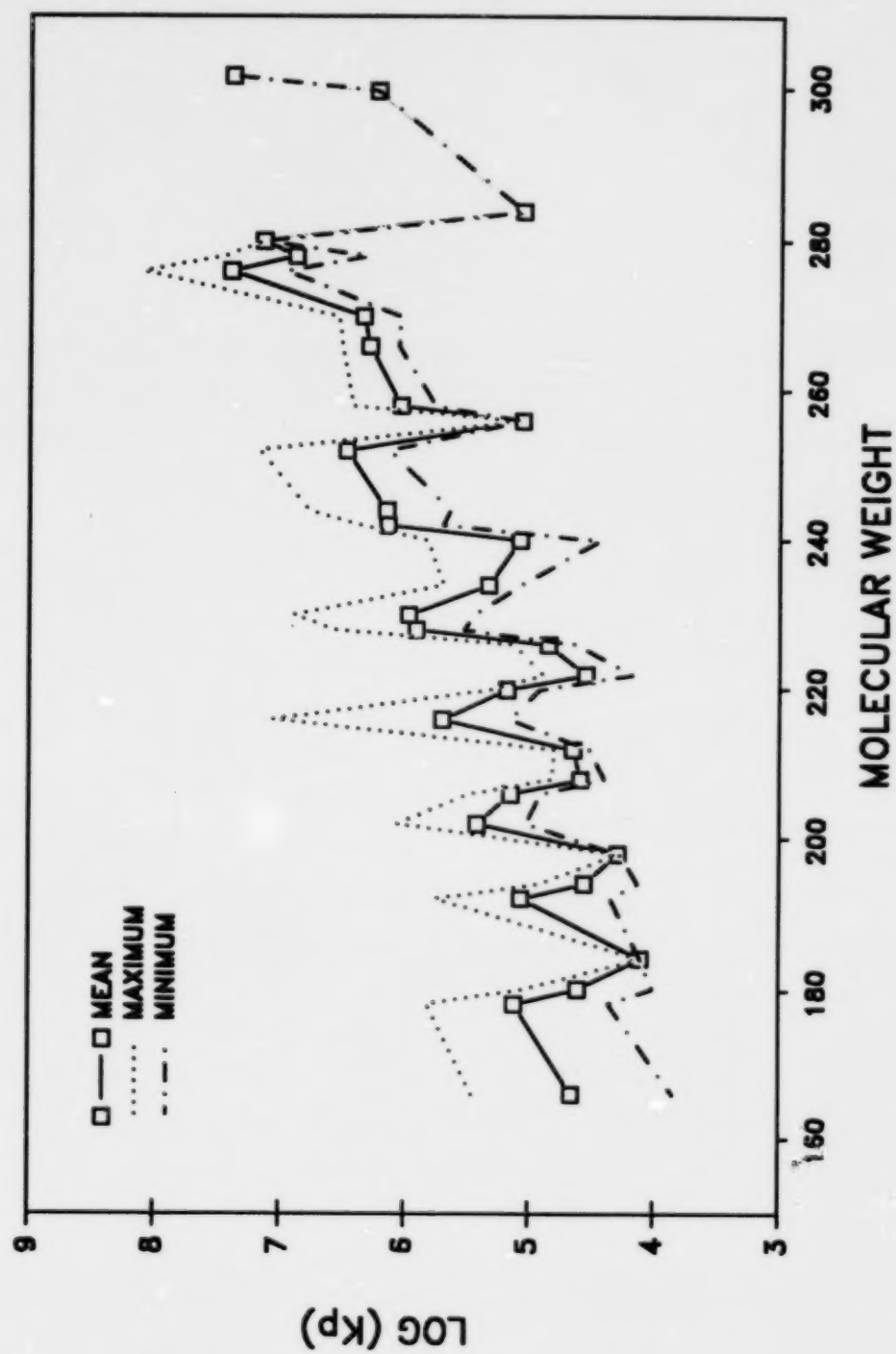


Figure 76. Partition coefficients of organic compounds versus molecular weight.

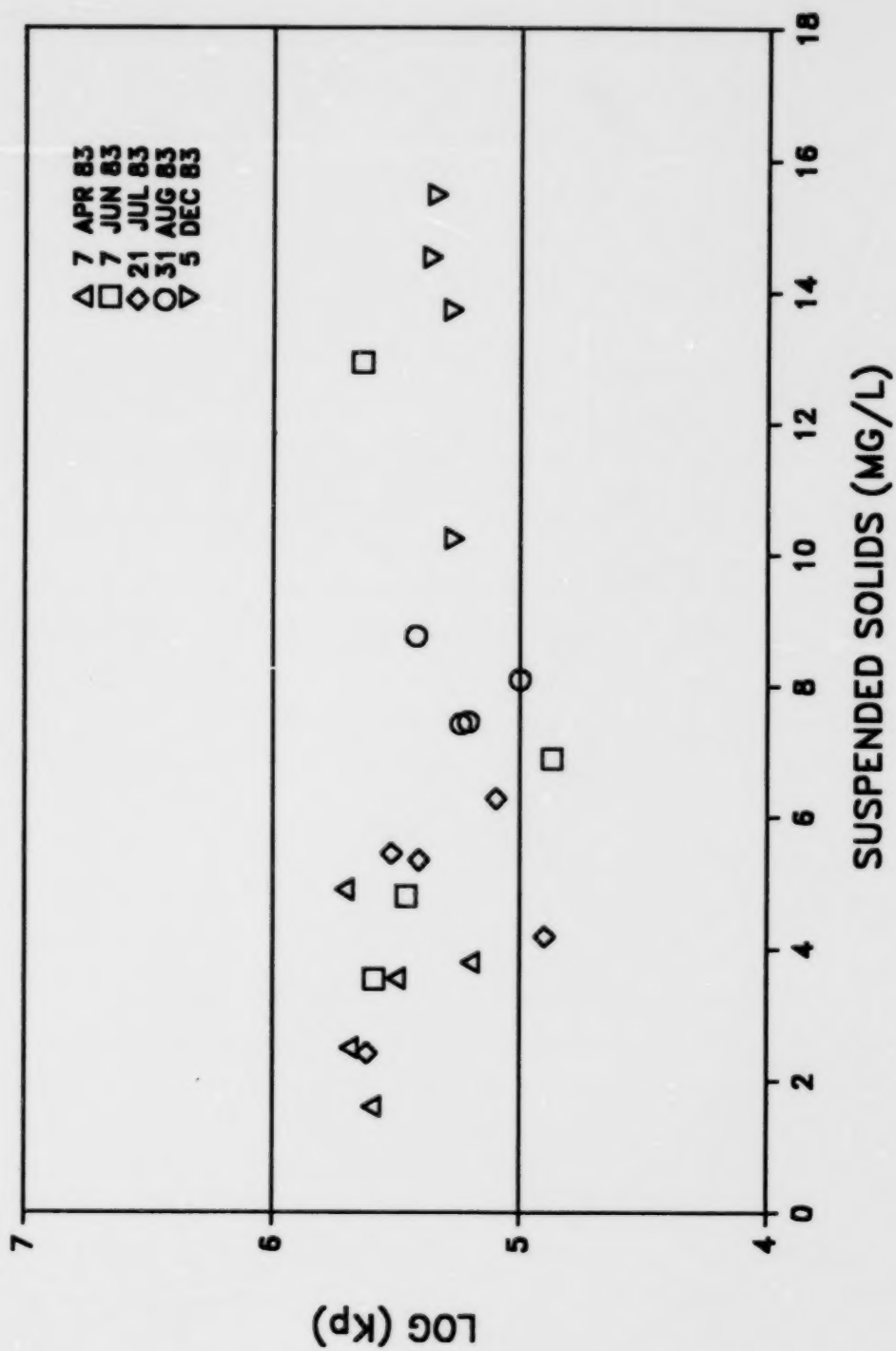


Figure 77. PCB partition coefficient versus suspended solids concentration.

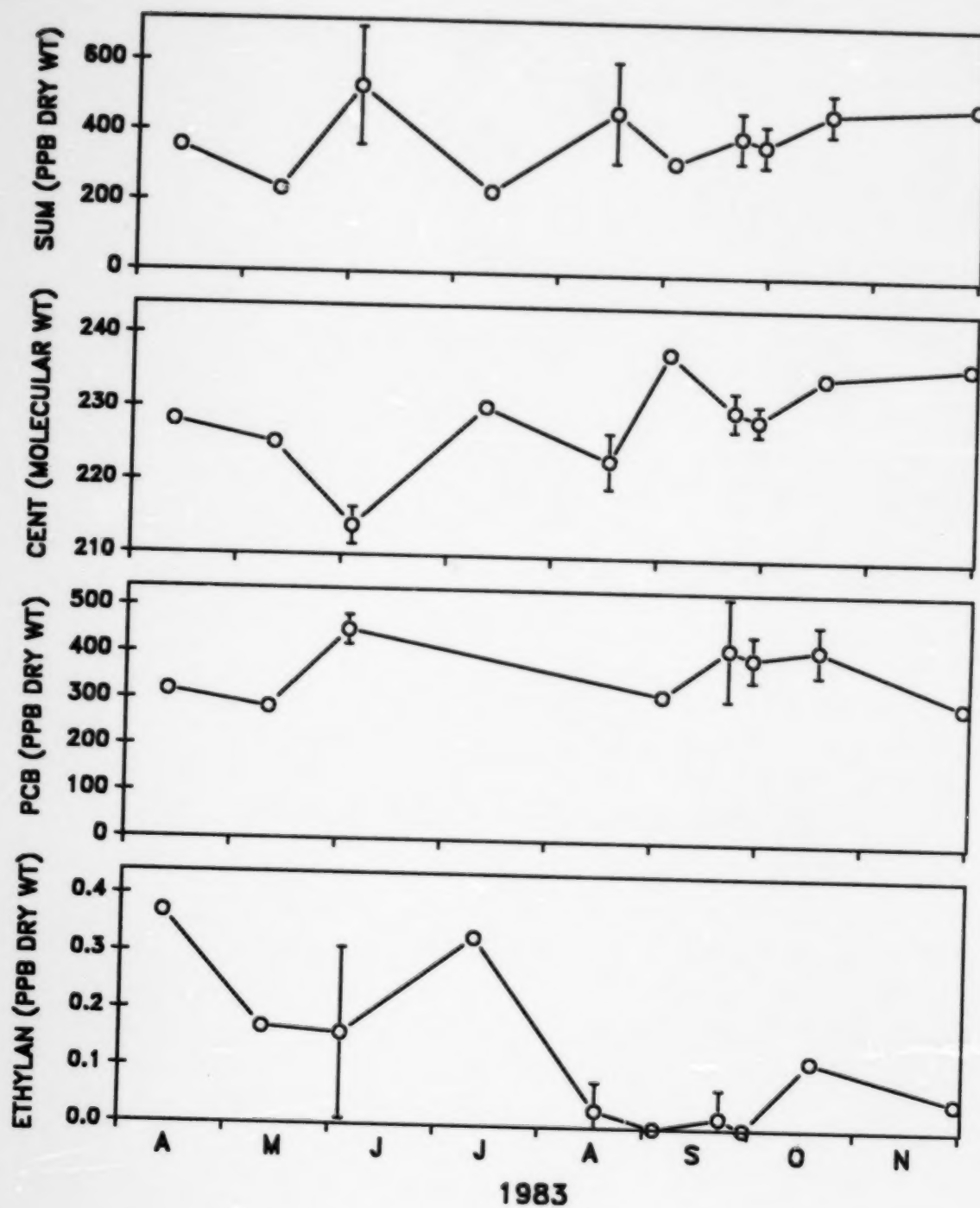


Figure 78. Organic contaminant tissue residues in Narragansett Bay reference mussels over time.

Farrington et al. (1983) in Narragansett Bay. Interestingly, low levels of Ethylan were also detected in Narragansett Bay mussels. However, these concentrations were near the detection limits of our analytical methods.

Organic tissue residues measured in mussels deployed prior to the disposal operation are given in Figure 79. PCB levels observed at this time agree with those found elsewhere in CLIS by Greig and Sennefelder (1985). Although statistical comparisons were not possible due to pseudoreplication of sampling units, some interesting differences in organic levels between stations were observed. Mussels deployed at CTR displayed nearly twice the level of total PAHs as did those deployed at the other biological effects stations. However, no clear differences in PCB concentrations were observed. Levels of both total PAHs and PCBs fell within the range of those observed in Narragansett Bay Mytilus determined in April and May 1983 (see Figure 78).

Levels of Ethylan were an order-of-magnitude greater in CLIS than those determined in Narragansett Bay mussels and, surprisingly, concentrations were highest at REFS. The route of exposure to this compound is unclear, as it was not found in predisposal sediment or water column samples from any station (see Tables 26 and 28). Additionally, order of magnitude elevations of the PAH Phenanthrene (PlA178; 37.5-206.1 ppb) were measured in deployed mussels compared with Narragansett Bay mussels (11.5 ppb in April). Tissue residues of this compound were highest at CTR, lowest at REFS, and intermediate at 400E and 1000E. The elevations of Ethylan and Phenanthrene suggest contamination of the water column prior to the disposal operation at the FVP site, possibly as a result of disposal operations occurring elsewhere in the CLIS disposal area.

Organic tissue residues measured in Mytilus retrieved

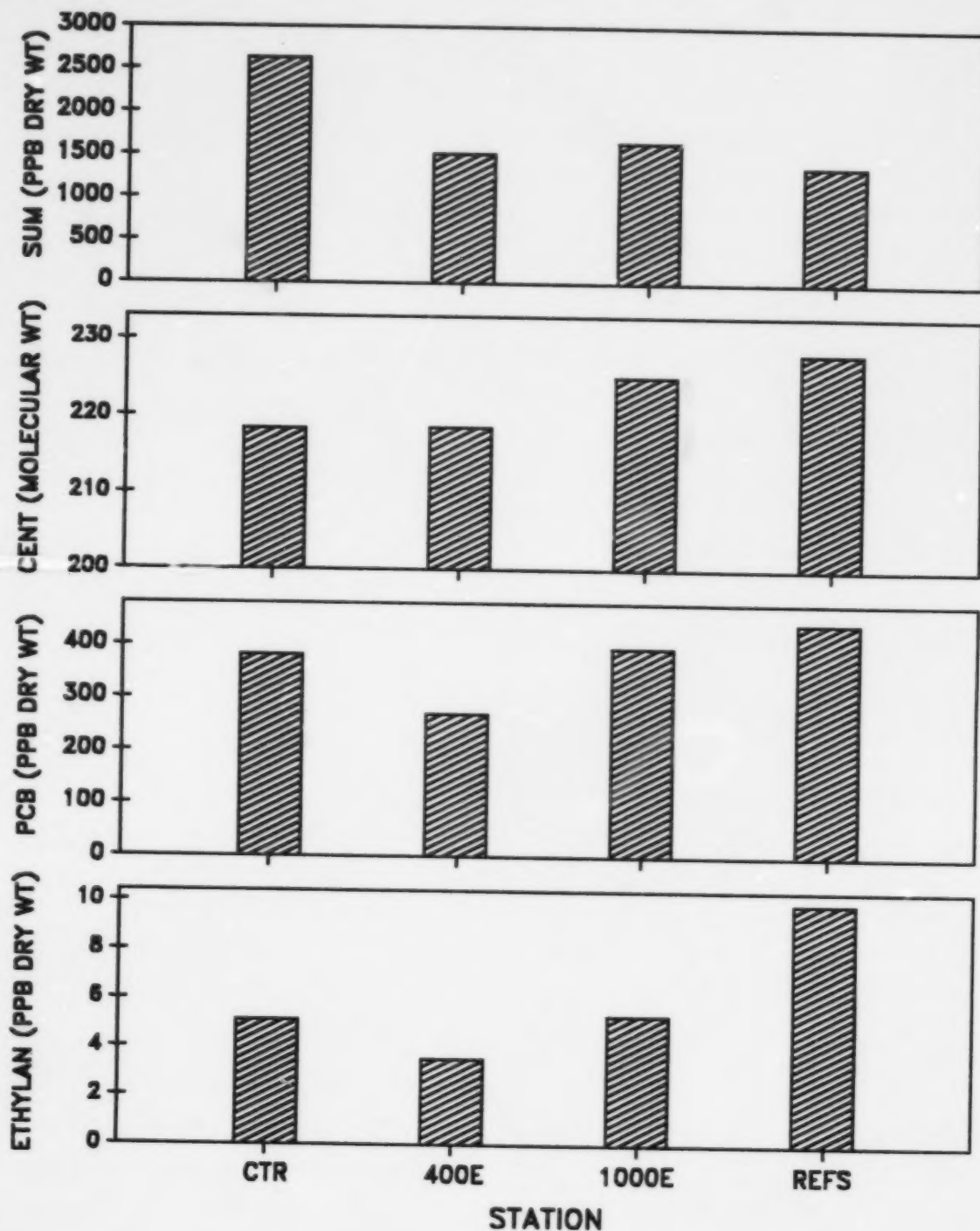


Figure 79. Organic contaminant tissue residues in predispositional mussels deployed at biological effects stations.

immediately following the disposal operation are given in Figure 80. Concentrations decreased along the measured gradient of exposure, such that levels were highest at 400E (mussels were not deployed at CTR during disposal) and lowest at REFS. Residues at all stations were elevated above both predisposal levels and those of the Narragansett Bay mussels. Following this 7 June sampling date, PAH tissue residues decreased rapidly such that by 13 July, concentrations had returned to predisposal levels (Figure 81). However, a trend could still be observed which corresponded to the gradient of exposure, suggesting continued release of contaminants from the disposal mound. Levels of PCBs and Ethylan at CTR, 400E, and 1000E were elevated above both REFS and predisposal concentrations. These elevations persisted into October 1983. The behavior of CENT during the entire study is somewhat difficult to interpret. The value of this summary statistic increased from low predisposal levels through July-August 1983, and remained essentially constant thereafter. This may reflect more rapid uptake of lower molecular weight PAHs, followed by rapid metabolism via mixed function oxidase (MFO) systems and/or faster depuration of these compounds. The initially low values of CENT at all stations may reflect predisposal contamination by low molecular weight PAHs.

Nephtys--

Organic contaminant tissue residues measured in native Nephtys incisa prior to the disposal operation in mid-April 1983 at 400E and REFS are given in Table 30. Levels of total PAHs and PCBs were 10 to 20% higher at 400E than at REFS. In addition, the value of CENT was slightly lower at 400E, indicating a distribution shifted towards low molecular weight PAHs. However, because statistical comparisons between stations could not be performed due to the method of collection and analysis of these samples, it was not possible to determine whether differences reflected low level exposure of 400E polychaetes to an array of organic contaminants similar to that found in BRH dredged

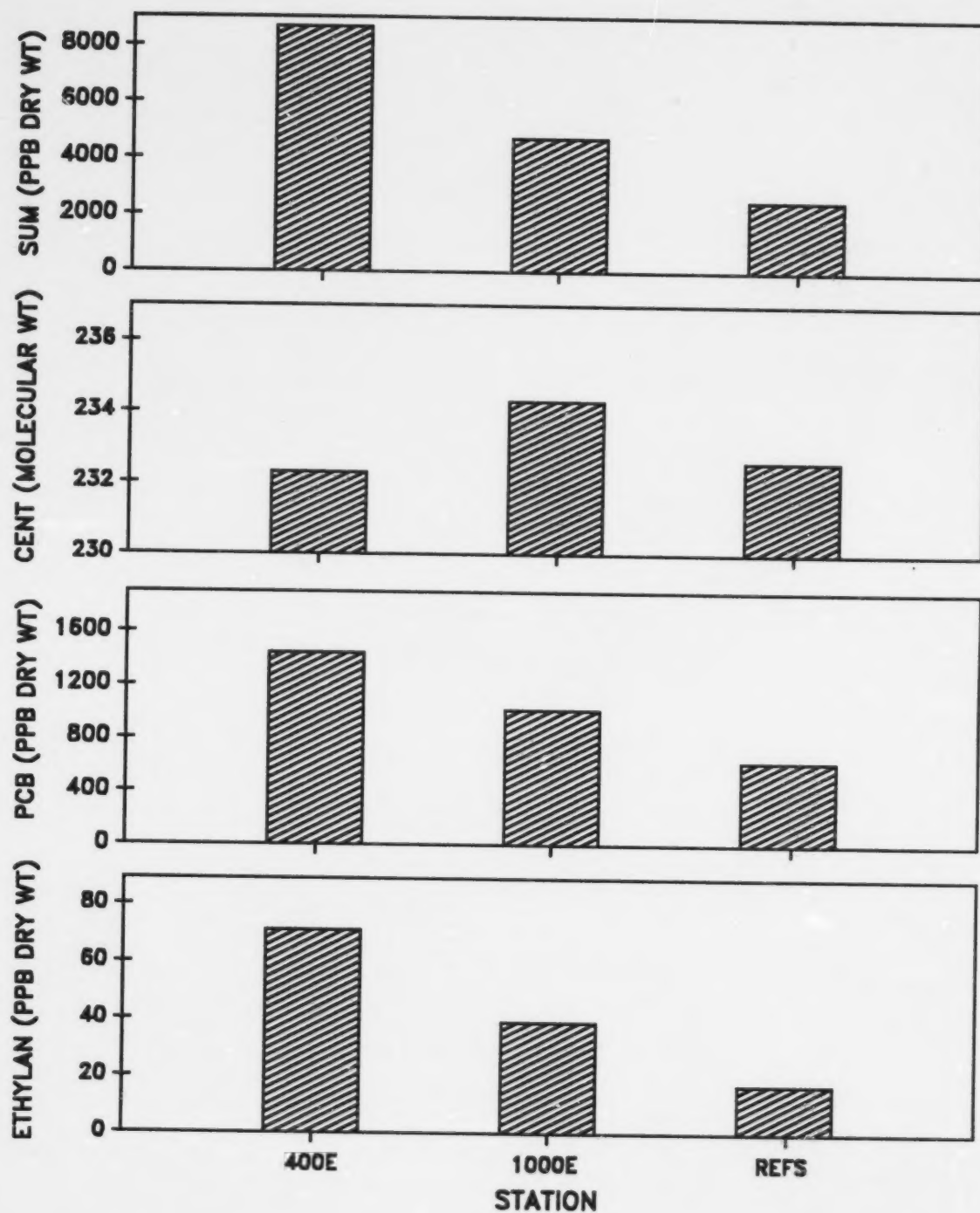


Figure 80. Organic contaminant tissue residues in postdisposal mussels deployed at biological effects stations.

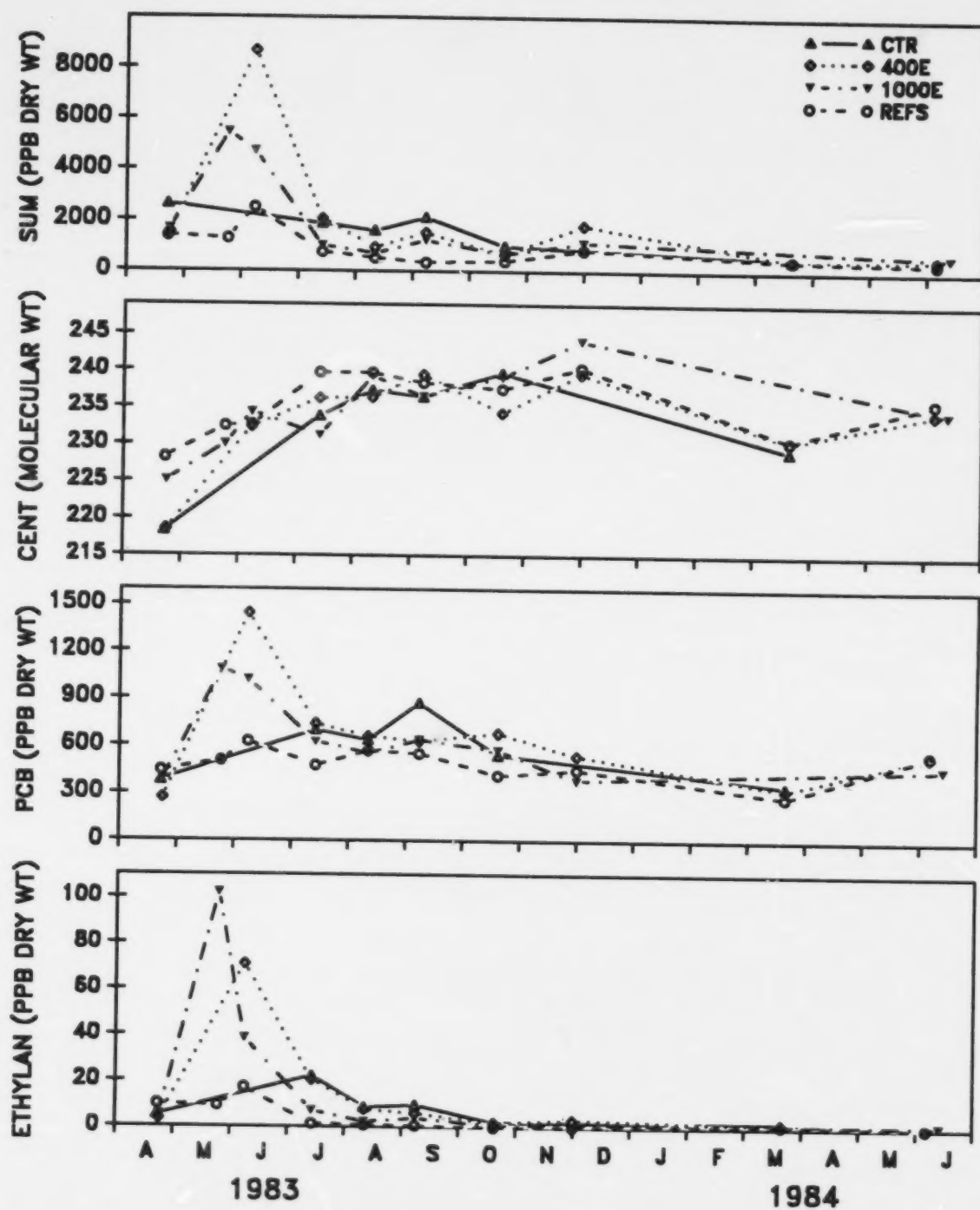


Figure 81. Organic contaminant tissue residues in mussels deployed at biological effects stations over time.

TABLE 30. PREDISPOSAL ORGANIC CONTAMINANT TISSUE
RESIDUES IN NEPHTYS INCISA

Contaminant	Station	
	400E	REFS
SUM*	962.8	711.0
CENT†	241.3	243.7
PCBs*	390.0	337.0
Ethylan*	0	0

* concentrations reported as ng/g

† units of molecular weight

material as opposed to sampling error. It is interesting to note that Nephtys collected two months earlier at these stations displayed similar differences. No Ethylan was detected in predisposal Nephtys from either station (Table 30).

Nephtys collected in early June displayed elevated levels of most organic contaminants at 400E and 1000E (infaunal organisms at CTR were buried by the disposal operation) relative to both predisposal concentrations and those at REFS (Figure 82). Elevations at 1000E again reflected contamination at that station as a result of water column transport from the mound and/or the nearby presence of dredged material from the presumed errant dump. The CENT statistic was depressed at 400E and 1000E in response to exposure to BRH material. Tissue residues at these two stations generally remained elevated relative to REFS throughout the study, although levels at 1000E had returned to predisposal conditions by March 1984. Ethylan was not detected in Nephtys collected from any of these three stations during the course of field work.

Nephtys was first collected from CTR in March 1984, and based upon their sizes, may have migrated to that site from surrounding areas. Thus the prior exposure history of these worms is uncertain. However, tissue residues were higher than those at other stations, and remained so through the last sampling period in October 1984. Ethylan was not detected in March, but was measured at relatively high levels in October (Figure 82). Throughout the study, a clear dose-response relationship existed among stations between organic tissue residues and contaminant concentrations in surficial sediments (compare Figures 74 and 82).

Summary

Levels of organic contaminants measured in sediment from

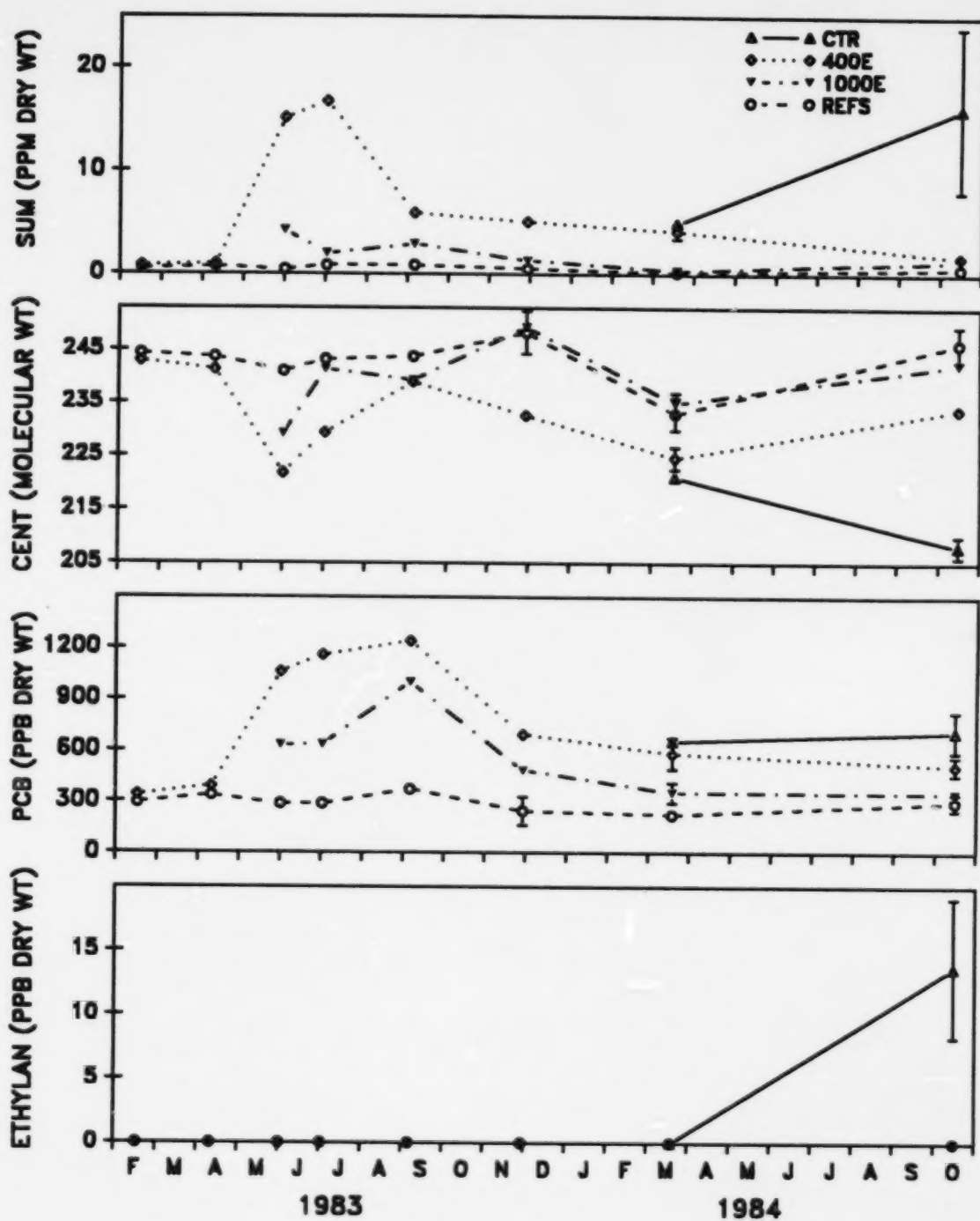


Figure 82. Organic contaminant tissue residues in Nephtys incisa from biological effects stations over time.

the FVP study reach were high and extremely variable. Mean PCB concentrations were 28 times those found in predisposal CLIS sediments. Concentrations of PCBs in BRH barrel material used in laboratory effects testing were three times the mean concentration in the study reach, although this difference may have been due to variation in analytical procedures used by NED and ERLN. Levels of other organic contaminants (PAHs, Ethylan) could not be compared as these were not measured by NED. However, based upon the results of trace metals analyses reported earlier, the BRH barrel material was assumed to be representative of the sediments dredged from the harbor. This barrel material contained high levels of both pyrogenic and petrogenic aromatic hydrocarbons (30 times background CLIS levels) and the chlorinated organic pesticide Ethylan (not detected in background sediments).

Sediments collected prior to disposal at stations in the immediate vicinity of the disposal site contained slightly higher levels of contaminants than did sediments from REFS, indicating residual contamination from historic disposal operations in CLIS. The disposal operation resulted in a gradient of organic contamination, with highest levels found at CTR and 200E. Concentrations of PCBs measured in surficial sediments immediately following disposal were approximately one third those determined in BRH barrel material, suggesting loss of contaminant mass during the dredging, barging, and disposal operations. Increased organic contamination was also observed at 1000E in a manner similar to that observed for trace metals, suggesting off-mound transport of BRH material during disposal and/or the occurrence of an errant dump nearby. Levels of PAHs, PCBs, and Ethylan remained elevated at all affected stations through late summer 1984, although physiochemical processes and dilution by clean, background sediments resuspended elsewhere resulted in apparent weathering of surficial sediments and lower contaminant levels. Organic contaminant levels in sediments at

REFS remained unaffected by the disposal operation.

Though poorly characterized by a limited number of samples, predisposal water column measurements indicated little definitive difference in contamination between CTR and REFS. Levels of all organic contaminants measured near bottom at CTR were elevated several fold one month following the disposal operation, but had returned to near-background levels two to three months later. Surface (5 MBS) samples also indicated some contamination at that site in June 1983. Weak organic signals were observed near bottom at 400E and 1000E when these stations were first sampled in early September 1983. The disposal operation (or concurrent operations elsewhere in CLIS) resulted in unexpected water column contamination at REFS, as short-lived elevations in PAH and PCB concentrations occurred immediately following disposal. Partition coefficients estimated from field samples agreed with those published for other aquatic systems, and appeared related to both molecular weight and molecular structure. No influence of suspended solids concentration on $\log(K_p)$ could be detected over the relatively narrow range of solids concentrations encountered during the study.

Tissue residues of Mytilus edulis collected from a reference population in Narragansett Bay and deployed one meter above the bottom were equivocal with regards to predisposal station differences. However, levels of total PAHs (highest at CTR) and Ethylan (highest at REFS) were greatly elevated relative to the Narragansett Bay population. In addition, levels of the PAH Phenanthrene were elevated above predeployment levels, displaying highest concentrations at CTR, and indicating potential contamination from disposal operations other than the FVP. The route of exposure to REFS mussels remains unclear, as abnormal levels of Phenanthrene were not observed in sediment or water column samples collected prior to exposure. PCB levels were fairly uniform across stations, and were measured at

concentrations similar to those of Narragansett Bay mussels. The disposal operation resulted in a gradient of tissue residue levels with highest concentrations measured at 400E (mussels were not deployed at CTR during this period). This effect lasted three to four months after cessation of disposal, reflecting water column contamination. Mussels at REFS displayed no apparent effect from the disposal operation.

Tissue residues of native Nephtys incisa collected prior to disposal were slightly elevated at 400E relative to those at REFS, reflecting predisposal differences in sediment chemistry. The presence of dredged material greatly amplified these differences, such that levels of PAHs in 400E worms were 20 to 30 times, and PCBs three times those of REFS Nephtys in June and July 1983. Tissue residues of Nephtys collected at 1000E were intermediate between the two during this period, reflecting a dose-response relationship with sediment concentrations. This pattern persisted over the 1.5-year, postdisposal period of sampling. Nephtys were collected at CTR for the first time in March 1984 and, because of their sizes, had apparently migrated from surrounding area. Irrespective of their uncertain exposure history, these worms displayed the highest residues of all stations on that date. Ethylan was detected only in worms collected at CTR in October 1984. REFS Nephtys were unaffected by the disposal operation.

TOTAL CARBON

The results of total carbon analyses of sediment samples are shown in Table 31. Results of duplicate total carbon analyses, in general, agreed within 10% of each other. Exceptions to this occurred in some samples from the CTR station. These contained small pieces of rock or grit (see discussion of sediment physical characteristics above) which may have been the cause of variability seen between duplicates, as

TABLE 31. TOTAL CARBON CONCENTRATIONS (MG/G DRY WEIGHT) IN SEDIMENT
CORES FROM REFS, CTR AND 200E, AND BRH BARREL SEDIMENT

Sediment	Depth in core (cm)	Sampling date		
		26 Jul 1983	19 Mar 1984	11 Sep 1984
REFS	0-2	24.4*	23.4	23.8
	2-4	22.2	22.5	21.6
	4-6	23.4	21.6	23.3
	6-8	22.4	23.0	22.9
	8-10	22.5	21.7	20.3
CTR	0-2	26.8	10.9	20.8
	2-4	22.6	25.2	24.2
	4-6	19.3	19.3	19.5
	6-8	15.6	20.9	34.1
	8-10	28.4	23.8	35.3
200E	0-2	41.5	26.0	24.6
	2-4	52.8	36.5	31.5
	4-6	46.9	53.5	34.4
	6-8	28.2	58.3	28.3
	8-10	21.7	70.3	23.6
BRH			76.4 ± 0.8**	

* means of duplicate analyses

** mean and standard error of 10 analyses

well as the low total carbon levels seen in these samples.

Total carbon concentrations in the three REFS sediment cores were fairly uniform with both depth and time, ranging from 20.3 to 24.4 mg/g. There was a slight tendency for levels to decrease with depth in the cores; however, these changes were small and not consistent. In contrast, total carbon concentrations in the three CTR cores were highly variable with no consistent trends with depth or time. These results showed a wide range of values (10.9-35.3 mg/g), but were much lower than those of samples obtained from the barrelled BRH material (76.4 ± 2.4 mg/g).

Total carbon concentrations in the 200E cores showed the widest range of concentrations (21.7-70.3 mg/g), as well as the highest concentrations of all field samples. Distinctly different trends were observed in each of the three cores analyzed from this station (Table 31). The core collected on 26 July 1983 had a maximum concentration in the 2-4 cm section, with concentrations decreasing below that depth to a minimum of 21.7 mg/g in the 8-10 cm section. In the 19 March 1984 core, concentrations increased with depth from 26.0 mg/g at the surface to 70.3 mg/g at a depth of 8-10 cm. This was the only core in which total carbon concentrations approached those of BRH barrel sediments (76.4 mg/g). The 11 September 1984 core at 200E showed a total carbon maximum of 34.4 mg/g in the middle (4-6 cm) section of the core; concentrations decreased both above and below this section. Results of two REMOTS® surveys in May 1983 indicated a layer of dredged material of between 4 and 10 cm thick near this station (Scott *et al.* 1984). Thus, some of the observed variability with depth between cores may have resulted from absolute differences in dredged sediment thickness. Variability in grain size may also have influenced total carbon levels.

In Figures 83, 84, and 85, organic contaminant concentrations are plotted against total carbon concentration for CTR and 200E. Figure 83 shows a significant correlation ($r=0.973$, $p<0.01$) between total carbon and PCBs at 200E but not at CTR ($r=-0.004$, $p>0.05$). The same was true for total carbon and Ethylan (Figure 84), with a significant correlation at 200E ($r=0.984$, $p<0.01$) but not at CTR ($r=-0.185$, $p>0.05$). The correlations of total carbon and PAHs were significant at both locations (Figure 85); however, a better correlation was again observed at 200E ($r=0.875$, $p<0.01$) than at CTR ($r=0.580$, $p<0.05$). Several investigators have shown that total organic carbon content strongly influences the sorption of organic compounds such as PCBs (Steen et al. 1978, Weber et al., 1983), DDT series compounds (Choi and Chen, 1976), and PAHs (Karickhoff et al., 1979, Means et al., 1980) to soils, sediments, and suspended particulate materials. Therefore, the strong correlations between organic contaminant (PCBs, Ethylan, and PAHs) concentrations and total carbon at the 200E station were not unexpected.

Reasons for the poor correlations between organic contaminant concentrations and total carbon at the CTR station are not known. Results from analyses of organic contaminants and trace metals have shown CTR to be very heterogeneous in these parameters. Also, gravity cores taken at this station showed the grain size structure of these sediments to be varied, with interlayerings of sand, darkly colored silt, and relic clay. Such inhomogeneities in sediment type (resulting from the order of disposal of material dredged from Black Rock Harbor), may explain the inconsistencies at this station. However, surficial sediments at 200E also displayed particle size effects from the disposal operation, so the roles played by these factors are unclear.

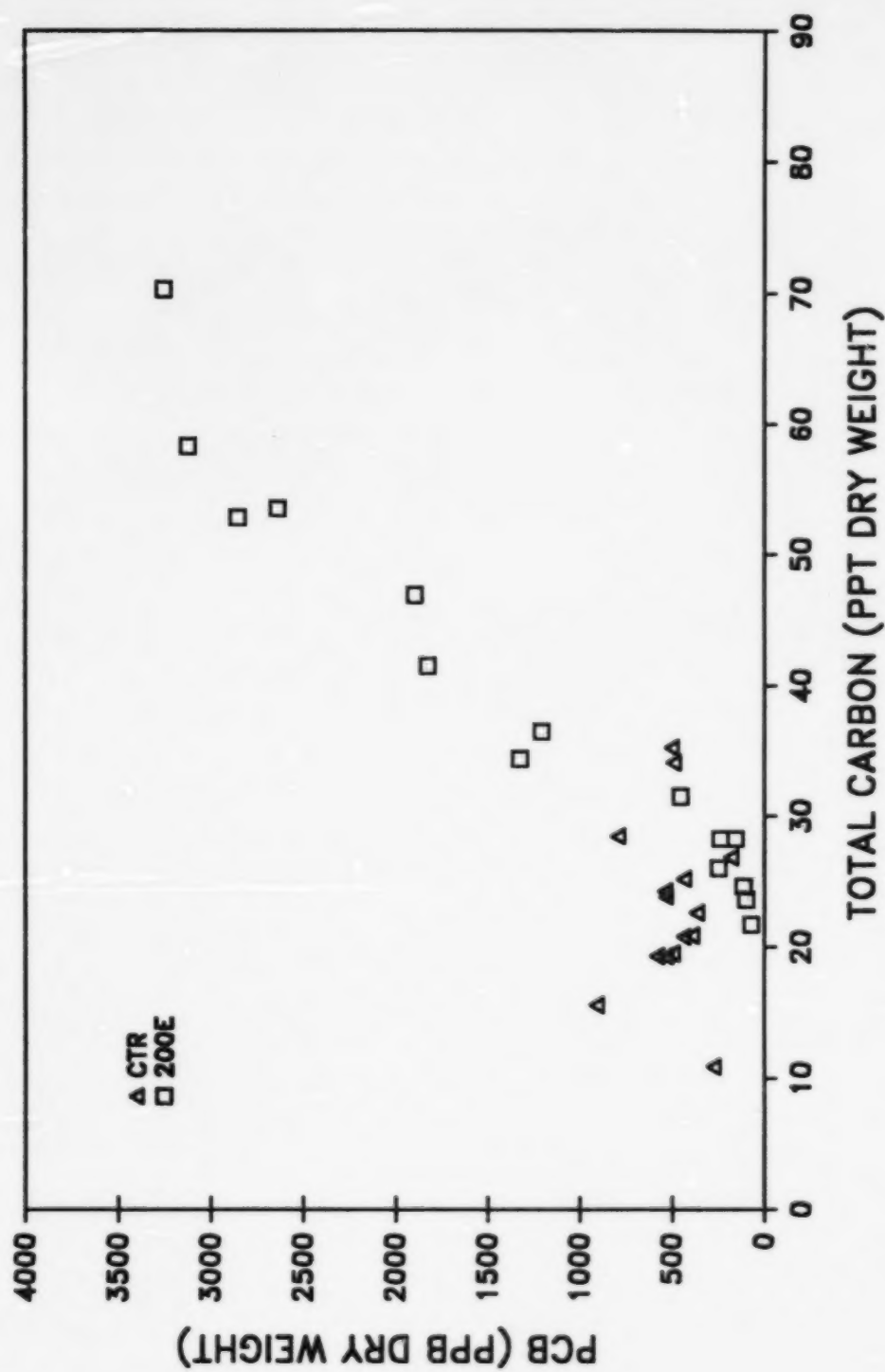


Figure 83. Concentration of Al254 versus total carbon concentrations in sediments from CTR and 200E.

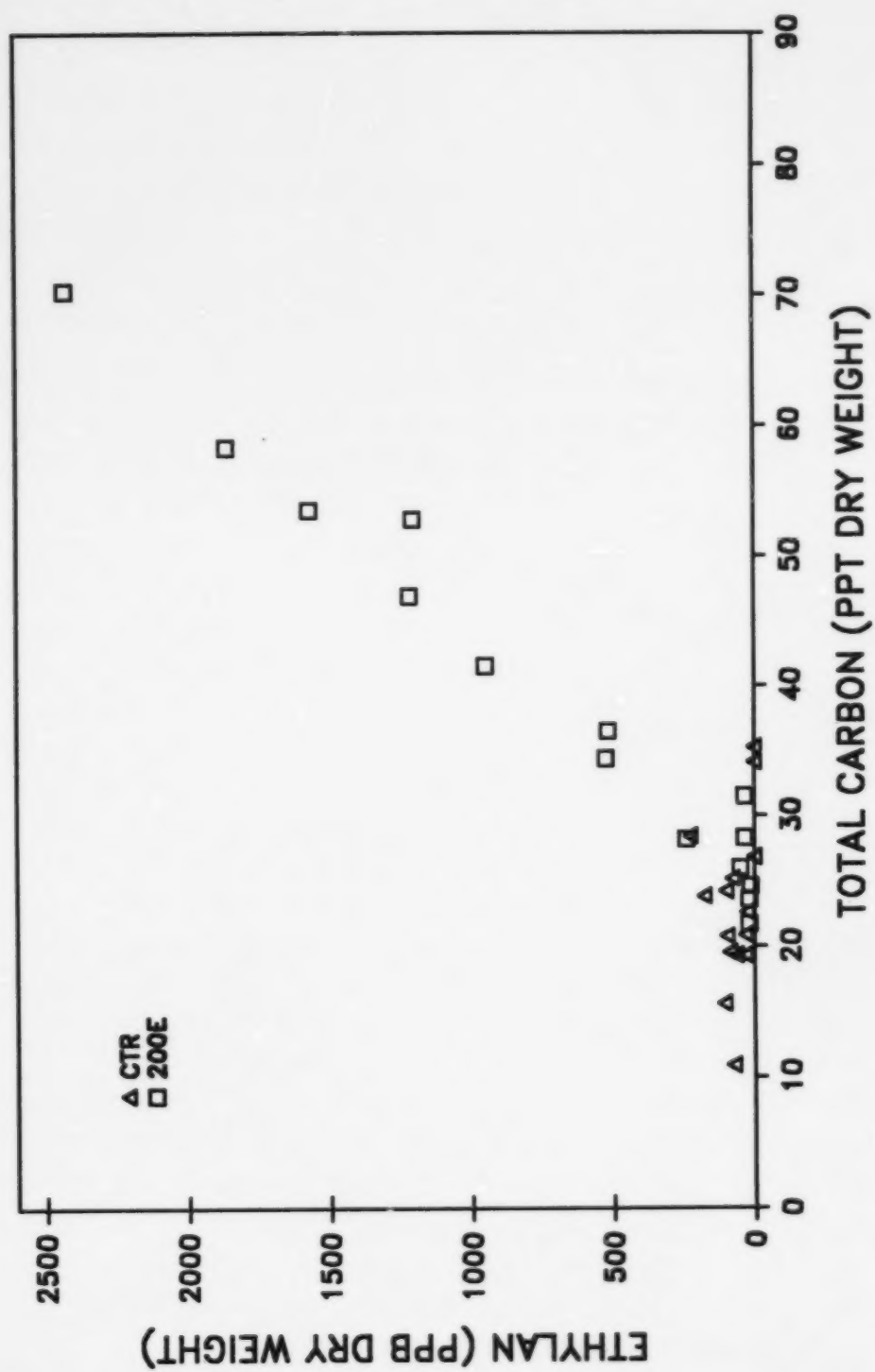


Figure 84. Concentrations of Ethylan versus total carbon concentrations in sediments from CTR and 200E.

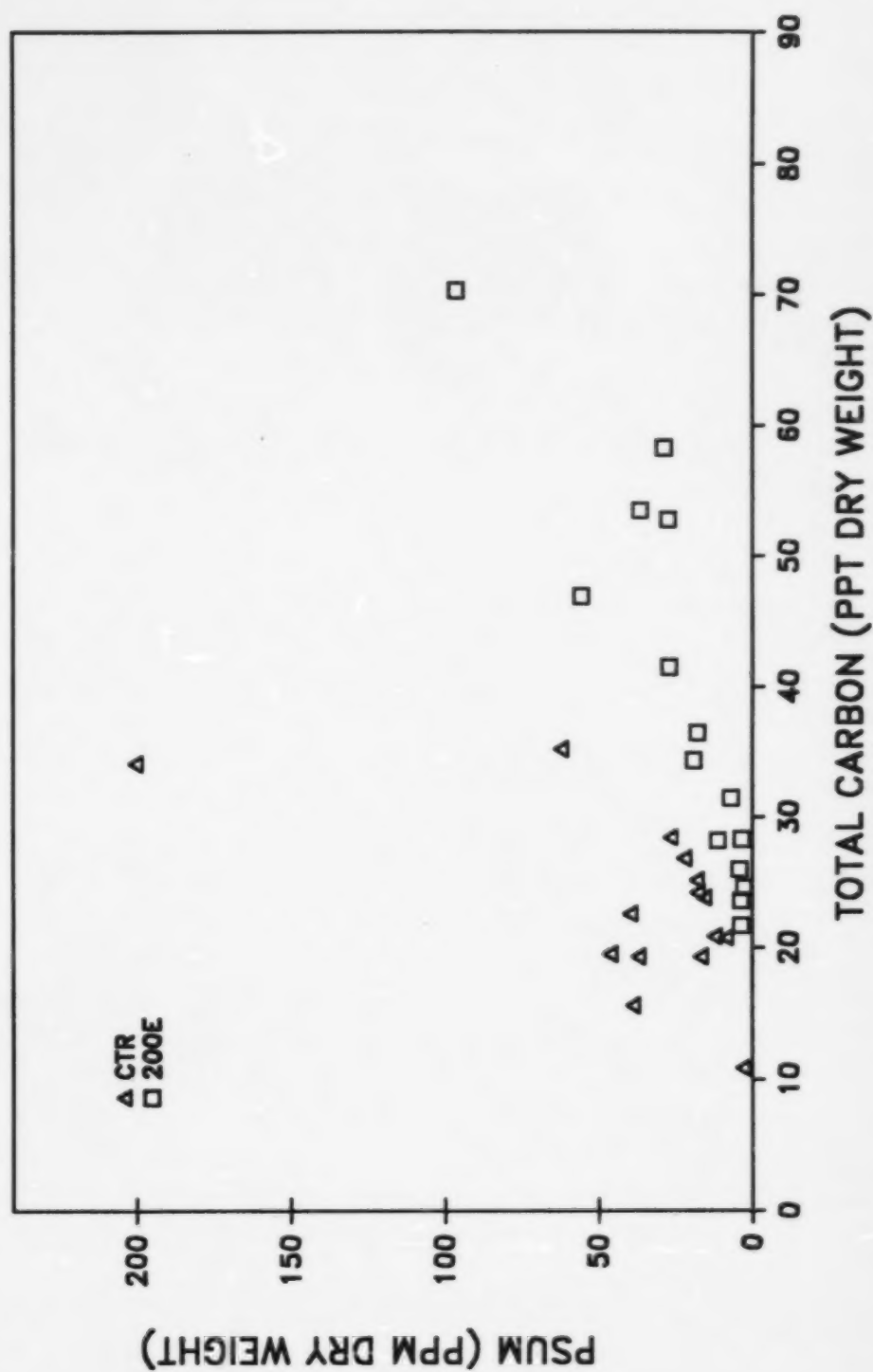


Figure 85. PSUM versus total carbon concentrations in sediments from CTR and 200E.

Summary

Total carbon concentrations in sediments from REFS were fairly uniform in both time and space, with values ranging from 20.3-24.4 mg/g. Carbon concentrations at CTR and 200E showed greater variability, ranging from 10.9 to 35.3 mg/g and 21.7 to 70.3 mg/g, respectively. A portion of this variability could be attributed to the distribution of dredged material at the FVP site. Maximum concentrations at both of these mound stations, however, were less than the mean carbon concentration found in barreled BRH material (76.4 ± 2.4 mg/g). Highest field concentrations (70.3 mg/g) were found in the 8-10 cm layer of the 200E sediment core collected on 19 March 1984, approximately 1 year following the disposal operation. The variability displayed at CTR and 200E is presumably related to the heterogeneous distribution of sediment particle sizes at these sites.

Total carbon concentrations at 200E were strongly correlated with PCBs, Ethylan and the PAH summary statistic PSUM. In contrast, carbon concentrations at CTR were correlated with PSUM only. Reasons for the general lack of correlation between total carbon and organic contaminants at CTR are poorly understood, although a partial explanation may lie in the large variation of contaminant levels and particle sizes found at this site.

SECTION 7

SUMMARY OF EXPOSURE ASSESSMENT FIELD DATA

The field program described in this report was designed in large part to address the first objective of the exposure assessment component of the FVP stated in the introductory remarks. Meeting that objective involved characterization of physical and chemical exposure conditions existing at the biological effects stations (CTR, 200E, 400E, 1000E and REFS) as the result of dredged material disposal. The following summary is organized around a comparison of environmental exposure conditions occurring at these five stations. In order to develop the field to laboratory link, that is, to simulate field conditions in the laboratory and thereby to establish the degree of correspondence between biological responses obtained in the laboratory with those seen in the field, the propriety of using barrelled BRH material in laboratory experiments as a surrogate for contaminated field sediments should be established. Although an examination of the events which took place during the disposal operation itself is beyond the scope of this discussion, the following summary of field information will therefore include comparisons between field sediments and those collected in BRH (and stored in barrels) prior to dredging.

The biological effects stations were initially established to represent a spectrum of exposure conditions (Phelps et al., in press). The four stations located at the disposal site itself were situated along a presumed gradient or transect of BRH contamination and influence. CTR, being located on the mound proper, represented a worst case exposure. Bedded sediments at

this station consisted solely of BRH material to a depth well below that normally ascribed to being within the zone of bioturbation. Resuspended BRH material was expected to produce highest levels of water column contaminants. The 200E and 400E stations represented lower exposures, as relatively thin layers of BRH material (generally 4-10 cm at 200E, a thin veneer at 400E) overlaid clean CLIS sediments; infaunal organisms could potentially move between these two sediment types during normal locomotor and feeding activities. Whereas water column conditions at 200E would be similar to those at CTR, conditions at 400E would be less contaminated than those at CTR due to dilution and repartitioning of BRH-associated contaminants with clean, background material resuspended from off-mound sites. The 1000E station was selected to be outside of the immediate influence of the mound. Water column contamination would indicate off-mound transport of BRH material. REFS was, of course, a chemical control station, well outside of the influence of disposal activities. Through combinations of BRH/reference sediment mixtures and BRH suspended solids concentrations, exposure conditions in the field could be replicated in laboratory effects experiments.

SEDIMENT EXPOSURE CONDITIONS

Prior to the disposal operation, physical and chemical characteristics of field sediments were fairly uniform. Particle size distributions displayed little variation between sediments collected at biological effects stations. Biological mixing depths were typically 3-5 cm throughout the area. Contaminant levels were also similar among transect stations, displaying low levels of PCBs (78.1 ± 7.0 $\mu\text{g/kg}$), total PAHs ($4,975 \pm 386$ $\mu\text{g/kg}$) and trace metals. The distribution of PAH compounds indicated these to have originated primarily as combustion products. There was, however, an indication of slightly elevated levels of organic contaminants at CTR, 200E

and 400E compared with predisposal REFS sediments, possibly reflecting historic disposal operations in the area. No Ethylan (a pesticide found in BRH material) was observed in predisposal sediments from any station.

Samples were not taken of barged BRH material actually disposed at the FVP site. However, sediment samples presumably representative of the disposed material (i.e. BRH barrel material) were strongly reducing and contained high levels of trace metals (e.g. $24.7 \pm 0.2 \mu\text{g/g}$ Cd, $2,900 \pm 122 \mu\text{g/g}$ Cu and $1,500 \pm 33 \mu\text{g/g}$ Cr), PCBs ($6.49 \pm 3.7 \mu\text{g/g}$), total PAHs ($146.4 \pm 13.0 \mu\text{g/g}$) and the pesticide Ethylan ($3.9 \pm 0.3 \mu\text{g/g}$). Physical characteristics of the barrel material appeared to differ from field sediments collected prior to disposal: median grain size (0.030 mm) was larger in the barrel material and there were less fine particles (81.0%). This comparison is somewhat tentative due to differences in the methods used to quantify these particle size distributions.

Sediments collected at CTR subsequent to disposal typically displayed among the highest field concentrations of BRH-associated contaminants. However, surficial CTR sediments collected immediately postdisposal were generally depressed in trace metals relative to stations located on the flanks of the mound (see Figures 49 and 54). In addition, sediments were more coarse at CTR than those on the flank (see Figure 38). When the effect of particle size (and therefore surface area available as sites of chemical 'attachment') was removed by dividing each metal concentration by the corresponding concentration of Fe (a metal displaying similar concentrations between BRH and reference sediments), levels at CTR were equal to or greater than those obtained at the flank stations. Particle sorting (differential transport) during the actual disposal operation, tidal winnowing of the mound apex following disposal (see Scott *et al.*, 1984), and enrichment of disposed sediments with clean

sand are possible causes of the observed particle size differences between CTR and other stations. Surficial sediments from mound stations were also less contaminated than the barrel material (Table 54), suggesting loss of up one half of the contaminant mass during dredging, barging, and disposal operations. In contrast, 73% of the volume of dredged material was accounted for through precision bathymetry and REMOTS® surveys (R.W. Morton, SAIC, personal communication). The apparent loss of fine particles, supported by in situ observations of increased frequency of turbidity plumes during the disposal operation, as well as observations of increased modal grain size made through REMOTS® benthic camera surveys (Scott et al., 1984, 1987), may account in part for the lower contaminant levels at CTR. Weathering of surficial sediments at this site was indicated by general decreases in contaminant concentrations and by shifts in the distribution of PAH compounds.

Sediments collected at CTR display extreme variability in chemical and physical characteristics. Contiguous 2 cm layers showed large differences in both trace metals and organic compounds. Visual inspection of gravity cores collected 2 years after disposal indicated fine, relatively contaminated material sandwiched between layers of apparently clean sand and relic clay clasts. Much of this variation was presumably due to the order of dredging and disposal of material from different sections of the FVP study reach, which displayed extreme heterogeneity in contaminant concentrations (see Appendix E), and to inclusion of relic sediments underlying contaminated material in the harbor.

The layer of BRH material at 200E as determined by chemical signal varied from 4 cm to greater than 10 cm, but generally was observed to be approximately 6 cm. Chemical characteristics of sediments within this layer were similar to, though generally more homogeneous than, those at CTR. In addition, particle size

distributions of sediments collected at the two stations were also similar. Surficial sediments at 200E also displayed decreased contamination with time.

BRH material observed at 400E formed a layer approximately 2-4 cm thick. Though elevated above background, this material generally displayed less than half the concentration of contaminants found at either CTR or 200E (see for example Figure 49). Physical characteristics of these sediments were similar to both predisposal conditions and those at sites located off the mound, although some enrichment by large particles was observed. One explanation for the reduced contaminant levels in 400E surficial sediments may be that BRH material transported to this station during and after disposal were diluted fairly rapidly with background sediments through bioturbation and physically-mediated mixing. Repartitioning of contaminants onto clean suspended particulates during transport may have also occurred.

Of interest are the elevated contaminant levels found at 1000E. For instance Ethylan, a compound not found in predisposal sediments, was measured in relatively high abundance after disposal (Figures 72 and 74). Increased contaminant levels were restricted primarily to surficial sediments, suggesting transport of contaminated sediments to this site. Although in situ transmissometers indicated the passage of turbidity plumes from the disposal operation, and therefore some transport of contaminated material from the mound may have occurred both during and after disposal, there appears to have been an errant dump 50 m northwest of this station sometime during or shortly following the disposal operation (Feng and DeGoursey, 1984).

In contrast to the patterns described above, sediment concentrations of trace metals and organic contaminants were uniformly low at REFS throughout the study (see Figures 52 and

73), and remained chemically unaffected by the disposal operation and resulting mound. Particle sizes also appeared unaffected relative to predisposal, background sediments. In these regards, REFS appears to have been an appropriate control station for sediment chemistry.

Sediment exposure conditions summarized in the above discussions are reflected to a large extent in the tissue residues of the infaunal polychaete Nephtys. Prior to recolonization of these worms at CTR (approximately 6 months following disposal), highest levels of organic contaminants were found in worms collected from 400E (Figure 82). Upon recolonization at CTR, tissue levels in animals collected at this station were higher than those seen in 400E worms on that date. Elevated contaminant levels were also observed in Nephtys collected from 1000E, reflecting the sediment contamination observed at that site. Worms collected from 400E and 1000E displayed decreasing contaminant levels with time, in a fashion similar to that displayed by surficial sediments at those stations. Elevated tissue residues were still observed 1.5 years after disposal, however. Worms collected from REFS displayed no changes in tissue residues during this study, and remained unaffected by BRH material. A more detailed analysis of Nephtys bioaccumulation of BRH-associated contaminants is given by Lake et al. (1987).

WATER COLUMN EXPOSURE CONDITIONS

Water column exposure conditions were uniform across stations prior to disposal, and in such hydrographic characteristics as temperature, salinity and DO, remained unaffected by the disposal operation and resulting mound. Large scale variability (seasonal cycles, vertical stratification) in these latter parameters was associated with seasonal meteorological patterns. Variability observed on smaller

temporal and spatial scales was associated with the movement of heterogeneous water masses through the study area. Trace metal concentrations measured over the course of a single day at CTR indicated the possible passage of a disposal plume from disposal operations elsewhere in CLIS. In general, however, variability between stations was no greater than that observed at any given station over the course of a single day. Interstation differences in water column chemical signals were difficult to quantify due to the effects of cyclic tidal resuspension of bottom sediments (and therefore particulate-associated changes in concentration).

Water column chemical signals observed at CTR after disposal were of short duration. PCB signals at 1 meter above bottom were 7 times background during the first postdisposal cruise (approximately one month following cessation of disposal), but had decreased considerably by the following cruise one month later. Other contaminants showed similar trends. This signal was strongest near bottom and primarily reflected the resuspension of unconsolidated sediments during and immediately following disposal (contaminants associated with fine particles left in suspension during disposal would most likely have been flushed from the area shortly following cessation of disposal). Results obtained from the acoustic profilometer suggest that suspended solids concentrations increased geometrically with decreased distance from the sediment surface. Total contaminant concentrations presumably increased in a similar fashion. Thus organisms living near or on the sediment surface (e.g. gastropods, crawling crustaceans) experienced contaminant levels much higher than those recorded at 1 meter above the bottom.

Although organic water chemistry was not followed at 200E, 400E and 1000E for most of the summer following disposal, near-bottom samples collected at the latter two sites in late

August and December 1983 displayed background levels of contaminants. Contamination of these sites by trace metals was not observed beyond that attributable to differences in suspended solids concentration.

The transport of particulate or dissolved contaminants in the water column to REFS was evidenced by elevated organic contaminant concentrations immediately following disposal at the FVP site. These signals were low (only up to 3-4 times background) and transient, and were most likely due to the disposal of over 10^6 cubic meters of dredged material which took place at other sites in the disposal area during that period.

Mytilus deployed 1 meter off the bottom displayed patterns of tissue residues similar to contaminant levels observed at that depth in the water column. Levels of PCBs and Ethylan, for instance, were elevated above background at CTR, 400E and 1000E for 2 to 3 months following disposal. The magnitude of this elevation decreased with increasing distance from CTR. It is interesting to note that increased levels of Ethylan and some PAHs (most notably Phenanthrene) were observed in mussels retrieved from all stations prior to disposal, as well as in those from REFS throughout the study (Figure 81). Though not observed in the water column, increased chemical concentrations may have resulted from disposal operations at other sites of the CLIS Disposal Area. Elevated levels of Ethylan were also observed in the Narragansett Bay reference mussels used in each deployment (see Appendix H), although these concentrations were generally less than those seen in the field animals. A more detailed analysis of Mytilus bioaccumulation of BRH-associated contaminants is given by Lake et al. (1987).

CODA

The descriptions above give evidence to the large degree of variability in exposure conditions seen at the FVP disposal site. Although much emphasis was placed upon homogenization of the barrel material to be used in laboratory simulations, sediments on the mound proved to be extremely heterogeneous. Clearly, simulation of this heterogeneity in laboratory experiments would not be appropriate, as it would lead to large differences in exposure conditions between replicate treatments.

The layers of BRH at 200E and 400E were generally more uniform internally than that at CTR. However, these sediments displayed a second form of variability: relatively thin layers of BRH overlaid clean, background sediments. Infaunal organisms active to depths below the layer of BRH would have been exposed to low contaminant levels in addition to the higher levels in surficial sediments. The amount of time spent in each layer would determine (in part) whether individuals experienced a "fine-grained" exposure or a "coarse-grained" exposure (sensu Levins, 1968). A fine-grained exposure would be experienced if the organism frequently moved between sediment types. Conditions to which the organism would be exposed can be thought of as being the average concentration. A coarse-grained exposure would be experienced if the organism spent most of its time in one layer or the other. In the extreme, a coarse-grained exposure would permit some degree of refuge from the BRH material, such that modification of behavioral patterns could result in almost complete avoidance of BRH-associated contaminants. Bulk chemistry measurements could lead to over or under-estimation of the actual exposure conditions. Ideally, laboratory simulation of field sediment exposures would take into account this spatial variability on the appropriate scales.

The time course of exposures in the field also displayed

variability that should be taken into account in laboratory-field comparisons. Whereas sediment exposures at mound and flank stations remained extremely elevated through the summer, fall, and winter following disposal, water column contaminant signals displayed relatively short-term elevations. In addition, movement of heterogeneous water masses through the study area, although rather poorly characterized in this study, presumably resulted in a time-varying exposure to contamination resulting from the disposal operation. Realistic simulation of water column exposures occurring in tidally-driven systems such as CLIS may be difficult to achieve.

Quantification of spatial and temporal variation in exposure conditions resulting from disposal of contaminated materials in marine systems has received little attention in the past. The results described in this report provide information which will better enable environmental scientists to elucidate the processes affecting contaminant transport and fate, and as a result, will permit more realistic simulation of exposure conditions in biological effects testing. To this end, syntheses of the FVP exposure assessment data into process-oriented models will appear in forthcoming articles and reports.

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APPENDIX A
ERLN EXPOSURE ASSESSMENT FIELD SAMPLES

WATER COLUMN

Key to sampling depth entries in tables:

- A = 1 meter above bottom
- B = 2 meters above bottom
- C = 3 meters above bottom
- D = 5 meters below surface
- M = miscellaneous depths other than above
- V = vertical profile

Physical Parameters

TABLE A-1. SAMPLE COVERAGE FOR SUSPENDED SOLIDS MEASUREMENTS

Station	Cruise date*								
	22 Apr 1983	23 May 1983	7 Jun 1983	21 Jul 1983	2 Sep 1983	5 Dec 1983	21 Mar 1984	12 Apr 1984	6 Jun 1984
REFS	AD	AD	AD	AD	AD	ABM		ACD	ACD
CTR	ACDM	AD	ACDM	ACD	ACD	ABCD			
200N						AC			
200E						AC			
200S						AC		ACD	
200W						AC			
400N						AC	ACD		ACD
400E	AD	A	ACD	ACD	ACD	AC	ACD	ACD	ACD
400S	AD	ACDM	ACD	ACD	ABCD	ABC			
400W	AD	AD	ACD	ACD	ACD	AC		ACD	ACD
600E	AD	AD	ACD	ACD	ACD	A			
800E						AC			
1000E	AD	AD	AD	AD	AD	A			
						AB			

* Dates listed are for Exposure Assessment physical parameters cruises only. Suspended solids concentrations were also obtained at stations and depths indicated for organic chemistry sample collection in Table A-5.

TABLE A-2. SAMPLE COVERAGE FOR LIGHT ATTENUATION VERTICAL PROFILES

Station	Cruise date						
	22 Apr 1983	23 May 1983	7 Jun 1983	21 Jul 1983	2 Sep 1983	5 Dec 1983	21 Mar 1984 6 Jun 1984
REFS	V	V	V	V	V	V	V
CTR	V	V	V	V	V	V	
200N						V	
200E						V	
200S						V	
200W						V	
400N	V	V	V	V	V	V	V
400E	V	V	V	V	V	V	V
400S	V	V	V	V	V	V	V
400W	V	V	V	V	V	V	
600E						V	
800E						V	
1000E	V	V	V	V	V	V	

TABLE A-3. SAMPLE COVERAGE FOR TEMPERATURE, CONDUCTIVITY AND SALINITY MEASUREMENTS

Station	Cruise date*							
	22 Apr 1983	23 May 1983	7 Jun 1983	21 Jun 1983	2 Sep 1983	5 Dec 1983	21 Mar 1984	12 Apr 1984 6 Jun 1984
REFS	V	V	V	V	V	V	V	V
CTR	V	V	V	V	V	V	V	V
200N								
200E								
200S								
200W								
400N	V	V	V	V				
400E	V	V	V	V				V
400S	V	V	V	V				V
400W	V	V	V	V				V
600E								
800E								
1000E	V	V	V	V				

* Dates listed are for Exposure Assessment physical parameters cruises only. Temperature, conductivity and salinity profiles were also determined for various stations during Effects Assessment cruises.

TABLE A-4. SAMPLE COVERAGE FOR DISSOLVED OXYGEN CONCENTRATION MEASUREMENTS

Station	Cruise date*					
	16 Feb 1983	12 Apr 1983	21 Jul 1983	2 Sep 1983	8 Sep 1983	9 Sep 1983
REFS	A	A	V	V		A
CTR				V		
400E	A	A	ABC	V	A	
1000E				V		

* Includes measurements made during Effects Assessment cruises.

Chemical Concentrations

TABLE A-5. SAMPLE COVERAGE FOR PCB AND PAH CONCENTRATION DETERMINATIONS

Station	Cruise date				
	7 Apr 1983	7 Jun 1983	21 Jul 1983	31 Aug 1983	5 Dec 1983
REFS	ABD	AD	AD	A	B
CTR	AD	AD	AD	A	B
400E				A	B
1000E				A	B

TABLE A-6. SAMPLE COVERAGE FOR TRACE METAL CONCENTRATION DETERMINATION

Station	Cruise date							
	22 Apr 1983	23 May 1983	7 Jun 1983	21 Jul 1983	2 Sep 1983	5 Dec 1983	21 Mar 1984	12 Apr 1984
REFS	AD	AD	AD	AD	AD	ABM		ACD
CTR	ACDM	AD	ACDM	ACD	ACD	ABCD		ACD
200N						AC		
200E						AC		ACD
200S						AC		
200W						AC		
400N	AD	A	ACD	ACD	ACD	AC	ACD	ACD
400E	AD	ACDM	ACD	ACD	ACD	AC	ACD	ACD
400S	AD	AD	ACD	ACD	ABCD	ABC		ACD
400W	AD	AD	ACD	ACD	ACD	AC		ACD
600E			ACD	ACD	ACD	A		
800E						AC		
1000E	AD	AD	AD	AD	AD	A		
						AB		

SEDIMENT

Key to sediment sample type entries in tables:

S = surface scoops from Smith-MacIntyre grab samples
 C = cores from Smith-MacIntyre grab samples
 G = gravity cores

Physical and Chemical Parameters

TABLE A-7. SAMPLE COVERAGE FOR SEDIMENT PHYSICAL AND CHEMICAL PARAMETER MEASUREMENT

Station	Cruise date									
	9 Dec 1982	2 Mar 1983	23 Apr 1983	3 Jun 1983	26 Jul 1983*	1 Sep 1983	9 Dec 1983	12 Mar 1984**	11 Jun 1984	20 Dec 1984
REFS	S	S								
CTR	S	S								C
200N			S	S	S	S	S	S	S	CG
200E	S	S								C
200S					S	S	S	S	S	
200W					S	S	S	S	S	
400N				S	S	S	S	S	S	
400E	S	S			S	S	S	S	S	C
400S				S	S	S	S	S	S	
400W				S	S	S	S	S	S	
600E										
800E										
1000E	S	S			S	C	C	C	C	C

* Cores from Smith-MacIntyre grabs were also collected on 26 July 1983 from 100N, 100W, 150N, 250S, 300N, 300E, 300W, 325S, 100E-100N, 100E-100S, 100W-100S, 100W-100N, 150E-150N, 250E-250N, 250E-250S, 300W-300S, and 300W-300N.

** Cores from Smith-MacIntyre grabs were also collected on 19 March 1984 from four locations approximately 50 m northwest of 1000E.

BIOTA

Tissue Contaminant Concentrations

TABLE A-8. DEPLOYMENT AND RETRIEVAL SCHEME OF CAGED MYTILUS EDULIS

Exposure condition	Stations	Deployment date	Retrieval date	Exposure duration
Predisposals	REFS, CTR 400E, 1000E	16 Mar 1985 16 Mar 1983	22 Apr 1983 23 Apr 1983	1 month 1 month
During disposal	REFS 1000E	23 Apr 1983 22 Apr 1983	24 May 1983 24 May 1983	1 month 1 month
During and immediate postdisposal	REFS 400E, 1000E	22 Apr 1983 23 Apr 1983	7 Jun 1983 7 Jun 1983	1.5 month 1.5 month
Two months postdisposal	REFS, CTR, 400E, 1000E	7 Jun 1983	13 Jul 1983	1 month
Three months postdisposal	REFS, CTR, 400E, 1000E	13 Jul 1983	10 Aug 1983	1 month
Four months postdisposal	REFS, CTR, 400E, 1000E	10 Aug 1983	6 Sep 1983	1 month
During and six months postdisposal	REFS, CTR, 400E, 1000E	16 Mar 1983	18 Oct 1983	7 month
Five months postdisposal	REFS, 400E, 1000E	6 Sep 1983	29 Nov 1983	3 month

(continued)

TABLE A-8 (continued).

Exposure condition	Stations	Deployment date	Retrieval date	Exposure duration
Eight months postdisposal	REFS, CTR, 400E	29 Nov 1983	20 Mar 1984	4 month
One year postdisposal	REFS, 400E, 1000E	12 Jun 1984	17 Oct 1984	4 month
Miscellaneous postdisposal	REFS, 400E	18 Oct 1983	5 Jun 1984	8 month
	REFS, 400E	20 Mar 1984	5 Jun 1984	2.5 month
	1000E	16 Mar 1983	12 Jun 1984	15 month
	1000E	18 Oct 1983	12 Jun 1984	8 month
	1000E	29 Nov 1983	12 Jun 1984	5.5 month
	400E	20 Mar 1984	17 Oct 1984	7 month
	REFS, 1000E	**	5 Jun 1984	

* Mussel cages moved from 400E to CTR on 7 Jun 1983.

** Indigenous population

TABLE A-9. SAMPLE COVERAGE OF INDIGENOUS NEPHTYS POPULATION

Station	Date									
	17 Oct 1982	16 Nov 1982	16 Feb 1983	12 Apr 1983	2 Jun 1983	3 Jul 1983	6 Sep 1983	29 Nov 1983	20 Mar 1984	16 Oct 1984
REFS	*	*	*	*	*	*	*	*	*	*
CTR										
400E	*	*	*	*	*	*	*	*	*	*
1000E					*	*	*	*	*	*